Assessing remotely sensed chlorophyll-a for the implementation of the Water Framework Directive in European perialpine lakes

Bresciani, M ; Stroppiana, D ; Odermatt, D ; Morabito, G ; Giardino, C

DOI: https://doi.org/10.1016/j.scitotenv.2011.05.001

Originally published at:
Bresciani, M; Stroppiana, D; Odermatt, D; Morabito, G; Giardino, C (2011). Assessing remotely sensed chlorophyll-a for the implementation of the Water Framework Directive in European perialpine lakes. Science of the Total Environment, 409(17):3083-3091.
DOI: https://doi.org/10.1016/j.scitotenv.2011.05.001

Assessing remotely sensed chlorophyll-a for the implementation of the Water Framework Directive in European perialpine lakes

Mariano Bresciani[a], Daniela Stroppiana[a], Daniel Odermatt[b], Giuseppe Morabito[c], Claudia Giardino[a,*]

[a] Optical Sensing Group, Institute for Electromagnetic Sensing of the Environment, National Research Council of Italy, via Bassini 15, I-20133 Milano, ITALY; bresciani.m@irea.cnr.it; stroppiana.d@iera.cnr.it; giardino.c@irea.cnr.it

[b] Remote Sensing Laboratories, University of Zurich, Zurich, Winterthurerstrasse 190, CH-8057 Zurich, SWITZERLAND; daniel.odermatt@geo.uzh.ch

[c] Institute for the Ecosystems Study, National Research Council of Italy, L.go Tonolli 50-52, I-28922 Verbania-Pallanza, ITALY; g.morabito@ise.cnr.it

[*] Corresponding author: Claudia Giardino, Optical Sensing Group, Institute for Electromagnetic Sensing of the Environment, National Research Council of Italy, via Bassini 15, I-20133 Milano, ITALY; e-giardino.c@irea.cnr.it; tel. +39-02-23699298, fax +39-02-23699300

Abstract

The lakes of the European perialpine region constitute a large water reservoir, which is threatened by the anthropogenic pressure altering water quality. The Water Framework Directive of the European Commission aims to protect water resources and monitoring is seen as an essential step for achieving this goal. Remote sensing can provide frequent data for large scale studies of water quality parameters such as chlorophyll-a (chl-a). In this work we use a dataset of maps of chl-a derived from over 200 MERIS (MEdium Resolution Imaging Spectrometer) satellite images for comparing water quality of 12 perialpine lakes in the period 2003-2009. Besides the different trophic levels of the lakes, results confirm that the seasonal variability of chl-a concentration is particularly pronounced during spring and autumn especially for the more eutrophic lakes. We show that relying on only one sample for the assessment of lake water quality during the season might lead to misleading results and erroneous assignments to quality classes. Time series MERIS data represents a suitable and cost-effective technology to fill this gap, depicting the dynamics of the surface waters of lakes in agreement with the evolution of natural phenomena.

Keywords: lakes; remote sensing; chlorophyll-a monitoring; Water Framework Directive

1. Introduction

Lentic ecosystems are inestimable renewable natural resources for biodiversity and can be significantly altered by human activities and climate change (Kaiblinger et al., 2009). Their ecological status vitally affects their value as drinking water reservoirs, for irrigation, fishery or recreation; any effort placed for preserving and/or improving the quality of these resources in the years to come and for increasing monitoring capability is justified. For this reason, the European Commission (EC) has adopted the Water Framework Directive (WFD) (Directive 2000/60/EC, 2000) with the main objective of maintaining 'good' and non-deteriorating status for all waters (surface, ground and coastal) (Chen et al., 2004). The Directive applies to all countries of the European Union (EU) and it has to be implemented at the catchment scale (river basin district) thus taking into account hydrological rather than geographical or political boundaries (Premazzi et al., 2003). Monitoring is an essential part of the implementation of the WFD (Premazzi et al., 2003). Surveillance monitoring aims at assessing the status of surface and ground waters. Operational monitoring is undertaken to evaluate the effects of measures undertaken to improve critical situations and to evaluate the level of achievement of the Directive's objectives; systematic monitoring has to be done through monitoring networks (Chen et al., 2004). So the starting step is the classification of water bodies into 'ecological quality status' using biological indicators (Kaiblinger et al., 2009). Lake classification under the WFD is based from the deviation of the present state from type-specific reference conditions (Wolfram et al., 2009). Moreover, the WFD forces the member states to monitor systematically all natural and artificial lakes with surface area ≥0.5 km2 (Premazzi et al., 2003). Field measurements of the indicators of lake water quality are to be carried out every three months although recent works suggest that the time interval between samplings should be reduced.

The sampling protocols adopted across EU suggest a minimum number of yearly samples for phytoplankton analysis. Sampling is usually carried out on a seasonal basis, therefore four samples is the most common option, although some Member States (such as Italy) decided to increase the minimum frequency up to 6 samples per year, trying to include those periods when phytoplankton is rapidly changing, due to the transition between two seasonal phases of relative stability in the assemblage composition. Because phytoplankton structure can be particularly variable from day to day during these transitional phases (Salmaso, 1996), the choice of the sampling dates is crucial, because it can significantly affect the outcome of the classification.

The number of samplings during the year is a compromise between the effort/cost required from the agencies in charge of monitoring and the need of describing the seasonal ecological variability with at least four to six sampling for all lakes. Within the time frames set by the Directive for measurements, the choice of the date is often dictated by the organizational needs hence the picture of water ecological conditions might be biased especially in the case of parameters which can vary significantly over small time periods, such as chlorophyll-a (chl-a).

The classification of the ecological status of lakes is based on the values measured for physicochemical and chemical parameters, which are related to the trophic level of the water: transparency, hypolimnetic oxygen, chlorophyll-a and total phosphorus. In the WFD chl-a concentration is therefore recognized as an essential parameter for the classification of the quality of lake waters (Jeppesen et al., 2003; Søndergaard et al., 2005) since together with phytoplankton species composition (Salmaso et al., 2006) is essential for an exhaustive description of the waters ecological status (Carvalho et al., 2008). In fact, these latter parameters are related to nutrient availability within waters (Vollenweider and Kerekes, 1980).

However, the estimation of chl-a concentrations in waters is often carried out with different methods for both sampling and laboratory analyses (Lindell et al., 1999) thus leading to the need of intercalibration of different monitoring systems as also required by the WFD (Nõges et al., 2010). Remote sensing technology is a tool for collecting consistent/homogeneous spatial and temporal data for the assessment of water quality parameters in lakes (e.g., Dekker et al., 2001; Koponen et al., 2004; Gons et al., 2008; Giardino et al., 2010). It allows us to monitor large water areas (from local to global scale) with systematic and cost-effective acquisitions. Salmaso and Mosello (2010) reviewed over 200 articles published in the field of limnology and draw the conclusion that synoptic analyses on a macro-regional scale in the sub alpine region need to be further addressed by researchers. Remote sensing can certainly provide the data to achieve this goal. Moreover, it offers archives of data which are suitable for the re-construction of historical trends (Chen et al., 2007). Several works have in particular focused on the satellite-inferred estimation of the concentration of chl-a (Baban, 1993; Giardino et al., 2001; Gons et al., 2002; Strömbeck and Pierson 2002; Doerffer and Schiller, 2007; Gitelson et al., 2007; Gower and King, 2007; Bilgehan et al., 2009; Bresciani et al., 2009). In fact, changes in the concentration of chl-a are associated to changes in the amount of water leaving radiance in the photosynthetically active radiation region of the electromagnetic spectrum and these changes can be detected with optical remote sensors (Baban, 1999). Therefore remote sensing can be a tool for collecting data to support of the implementation of the monitoring activities outlined in the EC WFD. However, despite the undisputable advantages offered by satellite sensors, remote sensing relies on the collection of in situ data for the development and the validation of algorithms and models (Chen et al., 2004). The systematic collection of in situ data should be performed as an integral part of operational monitoring build on remote sensing technology for lake management strategies such those as required by the WFD.

This work relies on the availability of a large dataset of maps of chl-a over 12 perialpine lakes for the period 2003-2009. Maps were derived from over 200 images acquired by the MEdium Resolution Imaging Spectrometer (MERIS) onboard the Envisat-1 mission of the European Space Agency (ESA) (Odermatt et al., 2010). The objectives are i) to compare water quality of the sub-alpine lakes of the study area by analysing chl-a concentrations, ii) to analyze the spatial variability of chl-a concentration during the season and over the years, and iii) to simulate the application of the WFD to this sub-alpine region through the information supplied by remotely sensed data.

2. The study area

The European perialpine region around the Alps comprises a significant number of glacial and tectonic lakes, which constitute an inestimable reservoir of freshwater. The 12 lakes selected in the study area (Fig. 1) belong to four countries: Italy, Switzerland, France and Germany. They have a size suitable for the geometric characteristics of the MERIS sensor (spatial full resolution is 300 m) and they are greater than 0.5 km^2, which is the minimum size required by the WFD for enforcing quality monitoring. Water of these lakes is extensively used for the production of hydroelectric power, is intensively used in agriculture and industry, becoming a life-sustaining element for the economy of the surrounding areas. Waters from the largest lakes (e.g., Lake Zurich and Lake Como) are even used directly as fresh water supply of the nearby cities. In addition, these water bodies are one of the key elements for the tourist economy of the Alpine region.

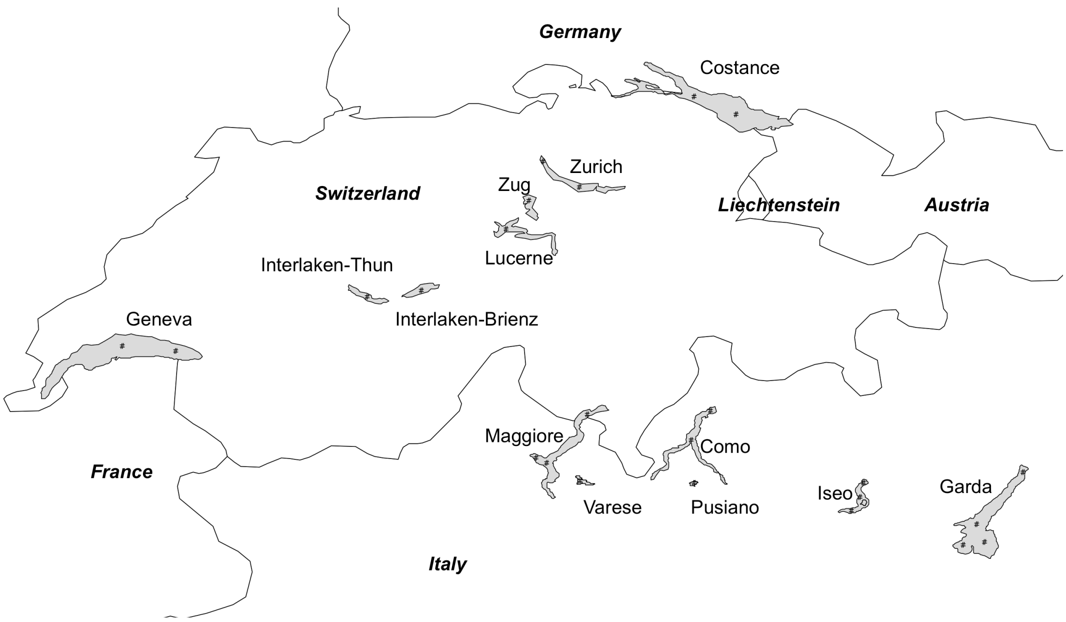

Figure 1: Study area with indication of Region of Interest (ROI) for every lakes.

The lakes included in this study are oligotrophic by nature (Guilizzoni et al., 1983) but they underwent intense processes of cultural eutrophication started in the 1960s, which reached a maximum at the end of the '70s. The major cause of the ecological deterioration was phosphorus loads from sewage, which triggered the increase of annual primary productivity, of algal biovolume and growth of cyanobacteria. Since the early 1980s, nutrient loads have been gradually reduced mainly by adopting treatment plants and by reducing the phosphorus contained in domestic detergents. Since then most of the lakes (i.e. the largest lakes) have started a re-oligotrophication process. Obviously each lake has had its own ecological history, which is extensively described in the scientific literature (e.g. Binelli et al., 1997; Ruggiu et al., 1998; Pugnetti et al., 2006; Anneville et al., 2007; Eckmann et al., 2007; Manca et al., 2007; Salmaso and Mosello, 2010; Stich et al., 2010).

The deep lakes of the Alps could be classified as warm monomictic lakes (e.g. Maggiore and Garda), the shallow lakes could be classified dimictic (e.g. Varese). Some common morphological features can be identified: the lakes are narrow and elongated, with steep sides and generally a flat bottom. The thalwegs of the largest Italian lakes are roughly oriented North-South whereas the others are mainly oriented East-West (Fig. 1). Except for Pusiano and Varese, which were included in the study for their peculiar trophic levels and their morphology, all lakes are characterized by high depth (both mean and maximum). The principal morphometric and hydrological features of the lakes are listed in Table 1. The lakes differentiate for their water circulation patterns, which lead to highly different residence times (0.7-26.8 years). In the table we also identify the lake typology (L-

Lake	Surface (km2)	Volume (km3)	Max depth (m)	Mean Depth (m)	Altitude (m a.s.l.)	Catchment area (km3)	Residence time (year)	Type
Como	145	22.5	410	154	198	4508	4.5	L-AL3
Constance	536	55	254	90	395	11500	4.3	L-AL3
Garda	370	50.35	346	136	65	2290	26.8	L-AL3
Geneva	582.4	89	310	154	372	7975	11.4	L-AL3
Interlaken	48.3	6.5	217	136	558	2500	1.85	L-AL3
Iseo	65.3	7.6	251	124	185	1777	4.2	L-AL3
Lucerne	113.6	11.8	214	104	434	2124	3.4	L-AL3
Maggiore	212	37	370	177	193	6599	4	L-AL3
Pusiano	5.2	0.07	27	14.5	257	94.3	0.7	L-AL4
Varese	14.5	0.1	26	11	238	112	1.8	L-AL4
Zug	38.3	3.2	198	83.2	413	204	14.7	L-AL3
Zurich	88.66	3.9	143	49	406	1829	1.2	L-AL3

Table 1: The major characteristics of the perialpine lakes object of this study. Lake typology ("Type") is assigned based on Wolfram et al. (2009).

AL3 and L-AL4), which is mainly assigned based on the above sea level (a.s.l) altitude and on the water mean depth as described by Wolfram et al. (2009): only lakes Pusiano and Varese are classified as L-AL4 lakes because of their shallow waters.

3. Materials and methods

The maps of chl-a concentration for the lakes of the study area were produced by Odermatt et al. (2010) from a dataset of over 200 MERIS images for the period 2003-2009. MERIS is a medium resolution imaging spectrometer carried onboard the ESA-Envisat-1 satellite and operating since 2002 within a mission covering open oceans, coastal zone and land surfaces. The instrument has a field of view around nadir of about 68.5° and it covers a swath width of 1150 km. MERIS is designed to acquire 15 spectral bands in the visible and near-infrared wavelengths with a spatial resolution of about 300 m at nadir.

All images were processed with the procedure available in the ESA Basic Envisat/ERS ATSR and MERIS (BEAM) that accounts for instrument smile, geometric, adjacency and atmospheric effects as described and validated in Odermatt et al. (2010). In particular, the adjacency effects were corrected with the Improved Contrast between Land and Ocean (ICOL) processor (Santer and Schmechtig, 2000; Santer and Zagolski, 2009). For the atmospheric correction and the retrieval of chl-a concentrations the Case-2 Regional (C2R) module (Doerffer and Schiller, 2008a; Doerffer and Schiller, 2008b), which implements a neural network for the inversion of both radiative transfer and bio-optical models, was used. C2R provided accurate reflectance values and consistent retrieval of chl-a concentrations in the study area (Odermatt et al., 2010). Outputs are given in the form of map products as well as in tabular form, whereas the latter allows the extraction of all relevant parameters and simple statistical variables for several sampling sites. Datasets were excluded i) if acquired at sun glint suspect geometries (i.e. above 10° east in summer or 20° east in winter), ii) if indicated by the C2R error flags, iii) if chl-a was below 0.1 mg/m3 or iv) if contrails or cirrus clouds were visible in a channel 13 quicklook.

For the definition of the sampling sites we first defined a set of Regions Of Interest (ROIs) of the size of 3 x 3 MERIS pixels over the lakes of the study area. For lakes with a surface area ≥0.5 km^2, the WFD locates the sampling point for monitoring water quality at the centre of the lake. This position is also ideal for optical measurements since it minimizes the adjacency effect, avoids mixed land/water pixels and ensures the least interference from the signal of the bottom of the lake. Therefore, we selected one ROI located at the centre of each lake of the study area. Moreover, for the largest lakes (surface area≥80 km^2) and for lakes of irregular shape, the WFD imposes additional sampling points to be defined case by case. For this reason we selected multiple ROIs for the largest lakes and we located them where the chl-a spatial variability might increase due to the presence of local factors (e.g. tributary water).

We extracted the temporal profiles of chl-a concentration for each ROI and analysed statistic metrics globally and for each season (spring, summer, autumn and winter) to compare the different trophic levels of the lakes. We chose to divide the year into the four seasons because hydrodynamic features and eutrophication are highly seasonal phenomena.

In order to achieve the third objective of this work (simulation of the application of the WFD with information supplied by remotely sensed data), we classified the lakes based on chl-a concentrations estimated for the seasonal periods given in Table 2. These periods are those suggested in the Italian official protocol for sampling lake phytoplankton, according to WFD (Buraschi et al., 2008). Within each season, we selected one date (Option A) mainly based on the quality and coverage of the satellite images. The ecological status classes were assigned based on reference values given by Wolfram et al. (2009). Since one major advantage of remote sensing is the availability of frequent observations at a low additional cost of acquisition, we applied the same classification by selecting a different date within the same periods of the year. Finally, we perform the same exercise by averaging all dates available from satellite images to simulate a condition of a full integration of remotely sensed data into a monitoring system of lake water quality as proposed by the WFD.

Season	**Reference range**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**

Winter	1 January-20 March	10/03	10/02	01/03	11/02	03/02	13/02	13/02
Spring	1 April-15 May	23/04	20/04	02/04	13/04	16/04	02/05	14/04
Summer	1 July-31 August	03/08	15/07	28/07	10/07	18/07	05/07	16/07
Autumn	1 October-31 November	28/10	24/10	29/10	10/10	15/10	09/10	25/10
Spring-summer transition	15 May-15 June	31/05	25/05	29/05	08/06	19/05	10/06	19/05
Summer-autumn transition	1 September-1 October	14/09	07/09	21/09	12/09	10/09	16/09	08/09

Table 2: The dates selected for each monitoring period defined by the Italian national protocol for sampling lake phytoplankton. If the MERIS acquisition were not available on the exact date shown here we chose the closest in time, which generally falls in an interval of ±3 days. These sampling periods correspond to Option A in Table 3.

4. Results and discussion

Fig. 2 shows some example maps of chl-a concentration over the largest lakes of the study area (i.e. Constance, Garda, Maggiore, Como and Geneva). The temporal trends observed in correspondence with the ROIs are presented in Fig. 3.

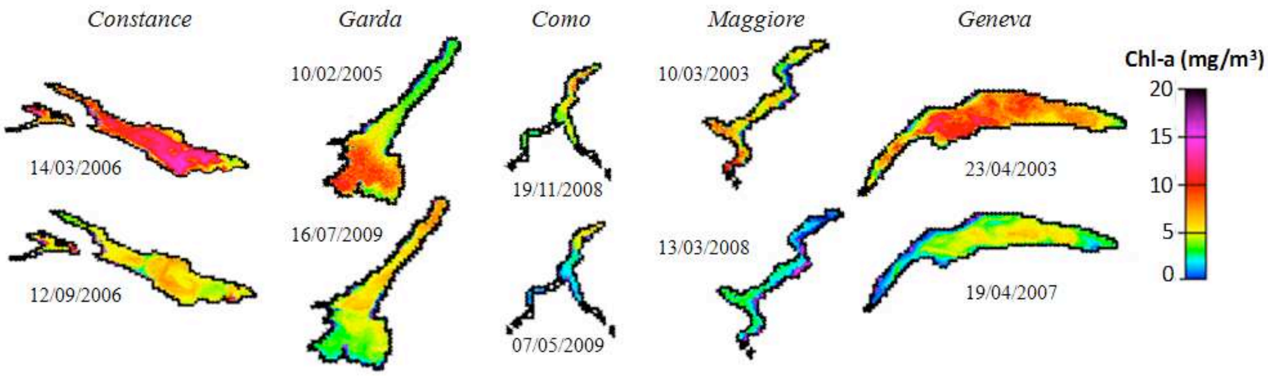

Figure 2: Some example maps of chl-a concentration over the largest lakes of the study area on key dates of the period 2003-2009.

Fig. 2 shows an algal bloom occurring in March and September 2006 in the central area of lake Constance where water is deepest. The map also shows the location of both the inflow (south east end) and outflow (west end) of the Rhein river as red spots of high concentration of chl-a (> 10 mg/m3). In fact, the difference of the profiles extracted for the ROIs is statistically different (**p<0.01). Indeed, the description of the spatial patterns of water quality parameters is one of the major contributions of remote sensing techniques for environmental monitoring. The oligotrophication of Lake Constance has begun at the beginning of this century as a consequence of the treatments of wastewater. This progressive process leads to low concentrations of chl-a and little fluctuations as shown by our results for the last seven years (Fig. 3). Moreover, our results highlight that in the very recent years and with the exception of the bloom observed in 2006, chl-a has stabilized below 5 mg/m3 confirming findings by Stich et al. (2010).

Chl-a concentration in Lake Como have decreased between 2003 and 2006 (Fig. 3) and have maintained stable after 2006: seasonal maxima have in fact decreased from above 10 mg/m3 to below 5 mg/m3 with a reduction of seasonal fluctuations. Moreover, the two sampled ROIs appear to have different values after 2006 although the difference is not statistically significant ($p=0.66$). This deep lake is geographically divided into three sub-basins (northern, western and eastern), which can be identified in Fig. 2; of these sub-basins, the deepest south-western Como branch is characterised by the lack of an outflow river and therefore by a longer water exchange time. By looking at the entire dataset, it appears rather difficult to identify regular, persistent and interpretable spatial patterns of chl-a concentration in Lake Como. The cause might be related to the atmospheric interference and quickly changing winds which modify the constituent patterns very frequently. The narrow y-shaped lake Como determines a behaviour similar to the typical behaviour of smaller lakes which are characterised by wide variability and unpredictability (Salmaso, 2010).

On the contrary, the size of Lake Garda leads to a minor susceptibility to the influence of hydrological and meteorological events hence more regular distribution of physical and chemical cycles (Salmaso, 2005). The map in Fig. 2 (16 July 2009) clearly shows the effect of inflow waters from the Sarca tributary at the northern edge of the lake with high concentration of chl-a (red colour key, chl-a ~ 10 mg/m3) while the effect of water mixing is clearly visible on 10 February 2005 (Fig. 2). The temporal trends collected over the three ROIs (Fig. 3) show a seasonal cycle with maxima generally occurring in spring. The difference between the northern ROI and each of the other two sampling points of the lake is statistically significant (***$p<0.001$): in fact, the northern region is

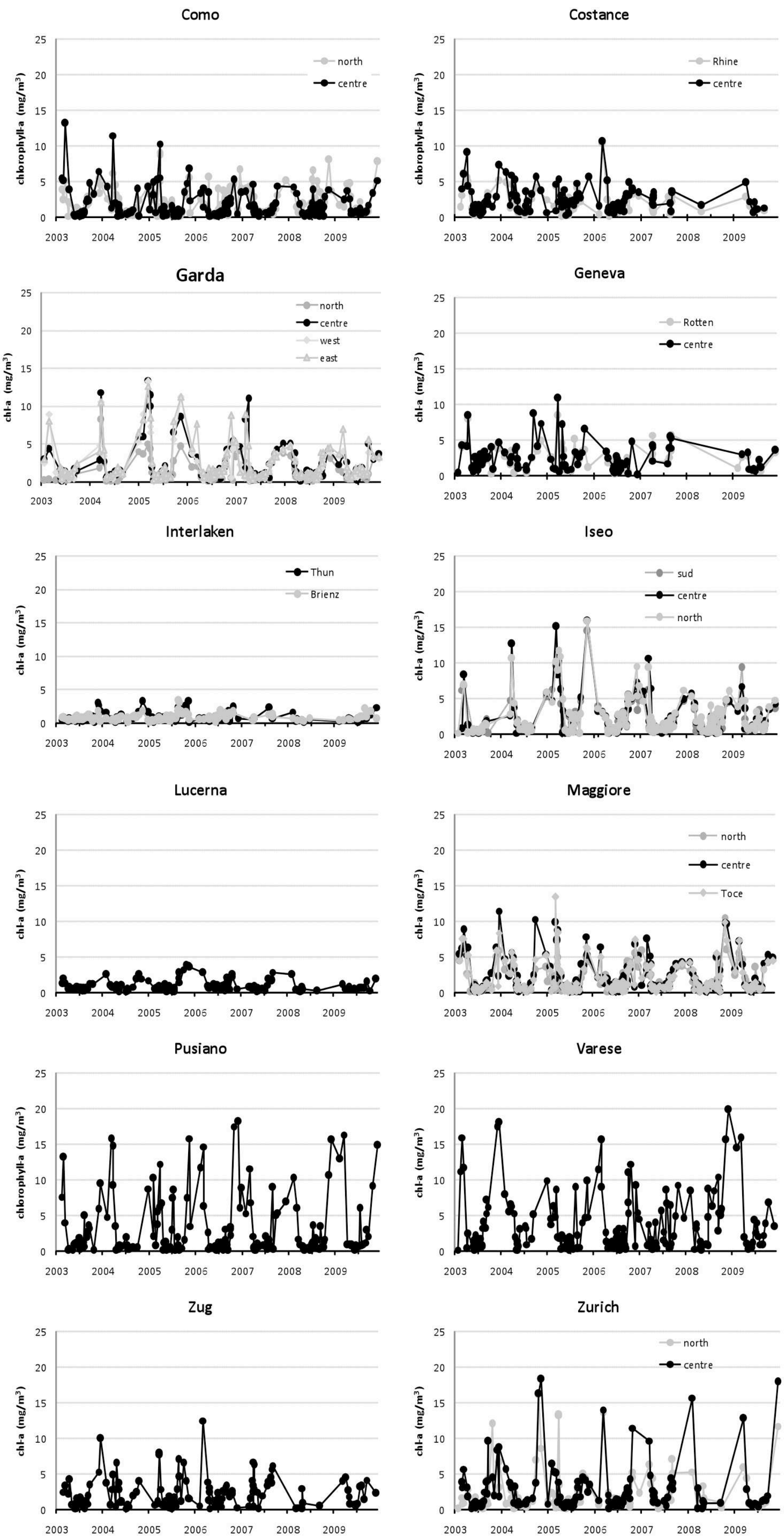

Figure 3: Trends of chl-a concentrations for the ROIs extracted for the lakes of the study area. The gaps within the time series are due to persistent cloud cover over the lakes or lack of data. The discontinuity affects in particular some of the lakes in 2008-2009.

characterised by lower concentrations with values below 5 mg/m3 whereas the other two regions have seasonal maxima above 10 mg/m3. This difference might be determined by several anthropic and geographical factors among which: the southern lake is the most densely populated and touristic area, the sewage collection is channelled towards the southern lake and winds favour the mixing of the shallower waters of the southern sub-basin.

Lake Maggiore shows a very high spatial variability of chl-a concentration from the lowest on one side to the highest values of the scale on the other side of the lake (i.e. west to east direction). This spatial variability is more comparable to the variability observed for Lake Como rather than Lake Garda as it would be expected for the similar dimensions. These changes might be the consequence of some residual adjacency effects (Odermatt et al., 2010) or they might be due to the effect of local meteorological conditions such as winds. Even the hydrodynamics of these basins could play a role, taking into account the strong differences in the residence times of the three lakes (Table 1).

In Lake Geneva, the largest of the study area, the spring algal bloom occurred in 2007 when in less than a week the concentration of chlorophyll a increased on average from 5 mg/m3 up to 10 mg/m3 and returned down to the initial values. The temporal trend (Fig. 4) highlight that maxima of chl-a concentrations (> 5 mg/m3) mainly occurred in either spring or autumn in the period 2003 to 2005, the spring peak in 2003 was also observed by Personnic et al. (2009); note that the lack of data in 2008 and 2009 might bias this results.

For the smaller lakes, we focused our analysis on the temporal trends (Fig. 3). Lakes Zurich, Zug and Luzern are geographically very close to each other covering an area about 100 km wide. However, Lake Lucerne is the most oligotrophic while lakes Zurich and Zug are mesotrophic to eutrophic. Our estimates confirm the greatest concentrations of chl-a in lake Zurich with seasonal maxima close to or above 15 mg/m3 and the lowest values, always below 5 mg/m3, in lake Luzern; in Lake Zug chl-a concentrations are generally below 5 mg/m3 but occasionally seasonal peaks can reach values of 10 mg/m3. Note that these peaks appear to be concentrated in the first three years of our analysis suggesting that this lake is still recovering from the severe eutrophication in the 1970s and 80s.

The Interlaken area comprises lakes Thun and Brienz, which are located at around 560 m a.s.l. (i.e. the highest lakes of the study area); many feeder rivers are of direct alpine origin (Kander, Lombach and Lütschine) and the ecological status is accordingly oligotrophic. Concentration of chl-a is very low (<3.5 mg/m3), with the lowest values estimated for Lake Brienz, and variations are minimal.

Finally, the smallest Italian lakes Iseo, Pusiano and Varese, all of them in eutrophic status, have the greatest chl-a concentrations and the highest level of fluctuation. In particular, in Lake Pusiano an algal bloom event occurs almost every year during 2003-2009 with chl-a concentrations above 15 mg/m3. These extreme conditions are mainly concentrated during either spring or autumn. The peaks of high chlorophyll a concentration of Lake Iseo are limited to sporadic events in winter compared to Pusiano and Varese.

We also computed the coefficient of variation ($CV=\sigma/\mu$) for each estimate within the 3 x 3 ROIs to evaluate the spatial variability of chl-a estimates. In the small lakes with high chl-a concentrations CV is grater than 15% for almost half of the dates, meaning that water quality parameters can vary also over small scales. As pictured in Fig. 2, the synoptic view of the sensors allows the description of the spatial variability of water quality parameters; this is indeed a major advantage offered by remote sensing techniques for the integration with field data collection, which, on the contrary, relies on the definition of representative sampling schemes.

The temporal trends do not show a tendency to either an increase or decrease of chl-a concentration. The regression lines have a slope not significantly different than zero and therefore the tendency is to maintain stable conditions. We computed the Hurst (H) exponent as an indicator of the expected tendency to confirm/reject the hypothesis of a stable behaviour. The results highlight that only in lakes Pusiano, Varese and Zug have high values close to 0.5 thus suggesting an un predictable behaviour.

The WFD foresees that the classification of the ecological status of a lake should be based on the annual average chl-a concentration derived from six seasonal samplings. We simulated the application of the WFD by using estimates of chl-a concentration derived from satellite images in correspondence with the central ROI of each lake and acquired on the dates shown in Table 2. Results are shown in Fig. 4 and compared to the limit set by the WFD between the classes “high” and “moderate” water quality (straight grey line in the graphs).

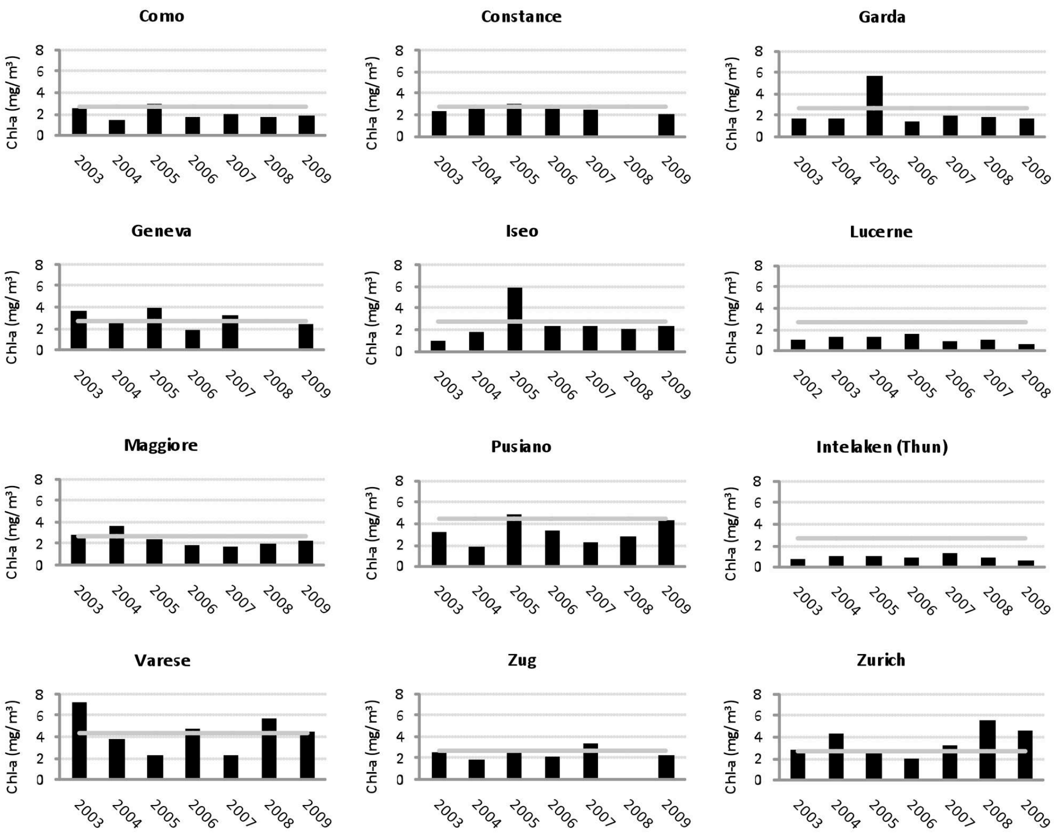

Figure 4: Chl-a concentrations derived from MERIS images (acquisition dates are given in Table 2) to support the application of the WFD. Values are the estimate of the central ROI for each lake. The straight line shows the limit between the classes high and good water quality as defined after the intercalibration exercise carried out inside the Alpine Geographic Intercalibration Group (Wolfram et al., 2009).

These results show that the chlorophyll concentration is often below the threshold established as boundary between high and good quality classes (as reported in Wolfram et al., 2009), even in lakes usually classified as meso- or eutrophic. In particular, lakes Lucerne and Thun are characterised by stable oligotrophic conditions (Friedrich et al., 1999; Finger et al., 2007). Among the most eutrophic lakes, only Lake Zurich is, in some year, classified as good and only in one situation (i.e.

2008) as moderate. However, Lake Zurich is mesotrophic as a consequence of the anthropogenic pressure along the shores and has had some significant events of cyanobacteria blooms (Peter et al., 2009). Lake Varese, in spite of his long history of water quality deterioration due to cultural eutrophication (Premazzi et al., 2003), is classified as good in most cases.

On the other side, the classification of lakes Garda and Iseo as moderate in 2005, derived from MERIS images, properly mirrors the exceptional worsening of water quality in 2005 caused by cyanobacteria blooms in the first (Salmaso, 2010) and by high nutrient loads, phosphorus in particular, in the second (Salmaso et al., 2007).

Since the WFD identifies only the large periods of the year when monitoring activity should be carried out, any value made available by satellite acquisitions within these periods is eligible for the classification. We therefore selected alternative dates to those given in Table 2 (hereafter named Option B). The results clearly show that this choice can be influential on the final classification especially in key seasons of the year when chl-a concentrations can significantly vary from day to day such as spring. Table 3 shows the example case of Lake Como where the ecological status assigned to the lake can change even from high to moderate/poor and vice versa.

	2003		**2004**		**2005**		**2006**		**2007**		**2008**		**2009**	
Season	A	B	A	B	A	B	A	B	A	B	A	B	A	B
Winter	5.13	5.38	4.22	4.22	0.72	4.90	3.31	4.16	3.63	3.44	4.29	3.39	2.46	2.46
Spring	3.81	3.81	1.90	11.4	10.3	5.58	0.96	3.47	1.27	4.67	0.50	1.78	3.78	2.63
Summer	0.45	0.20	0.38	1.27	0.59	0.32	0.41	0.28	0.65	0.65	0.55	0.55	0.36	0.71
Autumn	0.50	0.13	0.57	0.82	0.22	0.93	0.50	0.49	0.56	0.70	1.34	3.86	0.52	1.12
Spring-summer transition	2.43	4.71	1.08	1.08	0.73	0.80	2.50	1.44	1.58	1.18	2.11	0.78	0.62	0.78
Summer-autumn transition	3.22	6.47	0.17	3.93	4.81	6.83	2.33	5.26	4.43	1.93	1.37	2.81	3.36	5.21
Mean	2.59	3.45	1.38	3.80	2.90	3.23	1.67	2.52	2.02	2.09	1.70	2.20	1.85	2.15

Table 3: Chl-a concentration for the lake Como for the two Options (i.e. A and B in the table) of date selection made possible by the availability of frequent MERIS acquisitions during the periods outline by the WFD for monitoring lake water quality.

The high variability of chl-a concentration in spring is confirmed by the standard deviation of the estimates derived from satellite data available for each period shown in Fig. 5. In the same figure the standard deviation computed for the summer season is given for comparison. Spring is characterised by the highest variability and summer by the lowest due to stratification processes that reduce water flow between the strata. These dynamics are particularly effective in water resilience of the largest and deepest lakes (e.g. Garda). The most eutrophic lakes, such as Pusiano and Varese, are characterised by high variability of chl-a concentrations in both seasons. This would indicate that in the two lakes, summer chlorophyll concentration is less conservative than in the other lakes. The reason of this higher variability can be found in the differences in nutrient cycling across the trophic gradient. In eutrophic lakes, where the P supply at spring overturn is high, spring phytoplankton growth can produce high chl-a concentrations. The model suggested by Kufel (2001) can, probably, be applied also in our case: at the end of the growth phase, there is a strong nutrient flux towards the bottom of the lake, due to the sedimentation of decaying algal populations. This would remove a large amount of the nutrients from the upper water layers, thus limiting the phytoplankton growth. Under P limitation, a decline of chl-a is commonly observed in early summer in eutrophic lakes, until a new P input (i.e. from the metalimnion when the thermocline is deepening) refuels the algal population. On the other side, in the oligo- and mesotrophic lakes the spring growth is lower, therefore the downward P flux is less important and an higher amount of nutrients remain in the upper water layers and can be continuously recycled and exchanged between epilimnion and metalimnion: this would maintain a rather constant P availability, avoiding strong chl-a fluctuations during summer. The weaker temperature gradient in oligo- and mesotrophic lakes could enhance the probability of nutrient exchange between epilimnion and metalimnion.

The finding of higher variability in the most eutrophic systems seems to confirm the hypotheses made by Cottingham et al. (2000), who, by analysing paleoecological data, showed that nutrient enrichment results in much more marked fluctuations in chl-a concentration.

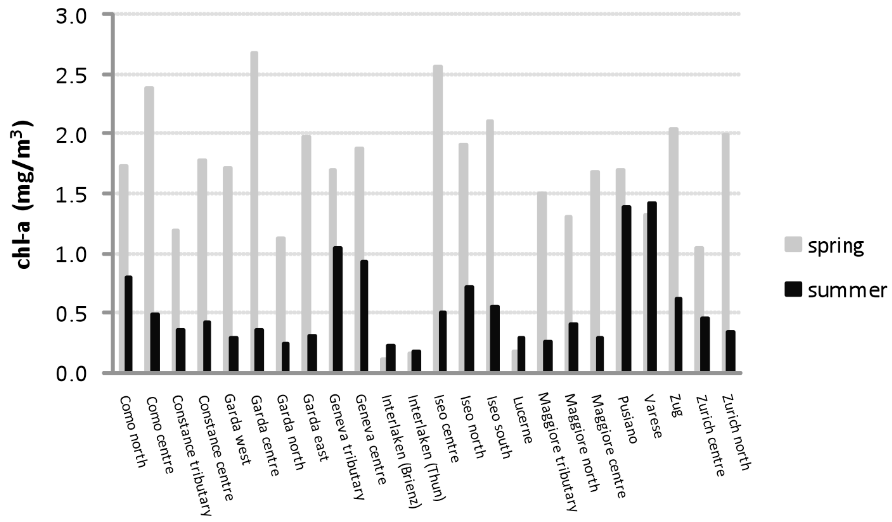

Figure 5: Standard deviation of the chl-a estimates derived from MERIS images available for spring and autumn seasons of the 2003-2009 period.

Fig. 6 shows the average, maximum and minimum values of chl-a concentrations estimated using all maps available for the six periods of the year outlined by the Italian sampling protocol. In Fig. 6 we compare these values to chl-a concentration assessed for the single date of Option A (Table 3). These results highlight that the high temporal dynamic of phytoplankton which should be captured by the monitoring system. In those lakes which have a significant variability one sampling date during the season might not describe accurately the trophic conditions of the lake. One sampling might in fact capture either negative or positive outliers; the former describe the individual extreme event while the latter provide reference good quality parameters. However, in both cases, the classification would be driven towards the erroneous quality class.

Besides the small lakes (Pusiano and Varese) and the case of lake Como already shown in Table 3, an example is lake Zurich where in 2006 the quality class varies depending on the use of either one date (Option A) or the average values.

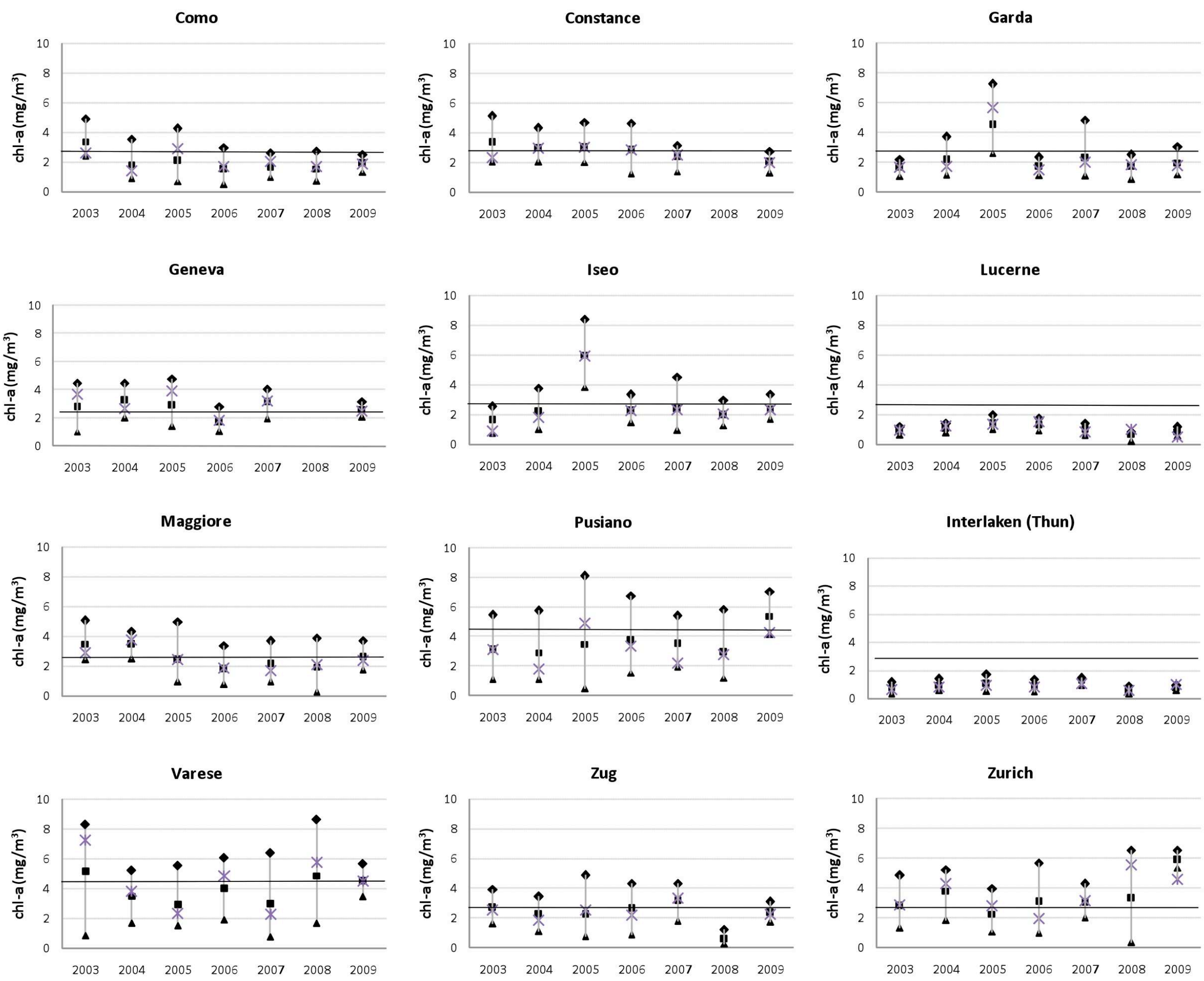

Figure 6: Average (squares), minimum (triangle) and maximum (rhombus) chl-a concentration in coincidence of the central ROIs derived from all product images available for the six key periods of the year. The estimates derived for the option A dates are shown for comparison with the cross markers. The straight line shows the limit between the classes high and good water quality as defined after the intercalibration exercise carried out inside the Alpine Geographic Intercalibration Group (Wolfram et al., 2009).

5. Conclusions

Environmental monitoring of surface waters can take advantage of remote sensing techniques which provide a synoptic view over large areas and frequent acquisitions. In situ data can provide a snapshot of local water conditions, which are however necessary for calibration and validation of satellite based models. Maps of chl-a concentration were obtained from processing more than 200 MERIS images over 12 perialpine lakes encompassing four countries for the seven year period 2003-2009. Results show the largest lakes (Constance, Garda, Como, Maggiore and Geneva) to be meso- to oligotrophic with occasional events of high chl-a concentration. These lakes show a clear seasonal trend of the concentrations with the highest values estimated in winter and during seasonal transitions from winter to spring and from autumn to winter; the lowest concentrations occur in summer. Luzern and Thun and Brienz, which form the Interlaken area, are the lakes with the best ecological conditions: the lowest chl-a concentration (<< 5 mg/m3) and the least fluctuations. On the other hand, the smaller Italian lakes, Iseo, Pusiano and Varese are characterised by high level of chl-a concentrations with peaks often above 10 mg/m3 and significant fluctuations in time. Despite the short time period of analysis, we observed that lakes are in stable conditions with the exception of Como, for which chl-a concentrations appear to have being decreasing.

We showed how remote sensing could be exploited for the implementation of the EU-WFD for classifying lake waters into quality classes. The small lakes are more frequently above the limit set for the high/good water quality class. On the contrary, larger lakes are in general high conditions with the exception of extreme events of algal blooms which temporarily worsen lake water quality during winter/spring. The use of remote sensing techniques for investigating phytoplankton abundance may overcome the problem of misclassification due to the chl-a seasonal variability and to the possibility of missing significant events when using the standard monitoring protocols with a low sampling frequency. On the other side, the vertical distribution of phytoplankton that in deep lakes is usually characterised by a metalimnetic chlorophyll maximum, often located around 10-15 meters depth, could have an impact on the remotely sensed signal (Stramska and Stramsky, 2005) so that during the stratification period the satellite-inferred estimates of chl-a might be lower than in situ measurements.

In order to assess water quality according to WFD criteria, the Members States monitoring programs suggest performing between four and six seasonal samplings during the year to average chl-a concentrations to identify a global quality class. We show that the assignment to a quality class can significantly depend on the date chosen for chl-a measurement/estimation since phytoplankton dynamics can vary from day to day. Since field campaigns significantly impact on the budget available for the implementation of the monitoring activities, remote sensing can be exploited to better describe these dynamics with multiple acquisitions with a lower marginal cost. If remotely sensed data are to be implemented in a monitoring system such as the one proposed by the WFD, research should focus on i) routine and extensive validation of the remotely sensed products through field data, ii) development of standard policies for satellite data acquisitions and criteria for interpreting products.

Acknowledgements

This study was co-funded by the EULAKES Project (EU Central Europe Programme 2010-2013). MERIS data were made available through the ESA projects AO-553 (MELINOS) and AO-1107. We are grateful to two anonymous reviewers for their useful comments on the manuscript.

References

Anneville O, Molinero JC, Souissi S, Balvay G, Gerdeaux D. Long-term changes in the copepod community of Lake Geneva. J Plankton Res 2007;29:49–59.

Baban MSJ. Detecting water quality parameters in Norfolk Broads, UK, using Landsat imagery. Int J Remote Sens 1993;14:1247–1267.

Baban MSJ. Use of remote sensing and geographical information systems in developing lake management strategies. Hydrobiologia 1999;395/396:211–226.

Bilgehan N, Karabork H. Ekercin S, Berktay A. Mapping chlorophyll-a through in-situ measurements and Terra ASTER satellite data. Environ Monit Assess 2009;157:375–382.

Binelli A, Provini A, Galassi S. Trophic modifications in Lake Como (N. Italy) caused by the Zebra Mussel (Dreissena Plymorpha). Water Air Soil Poll 1997;99:633–640.

Brando VE, Dekker AG. Satellite hyperspectral remote sensing for estimating estuarine and coastal water quality. IEEE T Geosci Remote 2003;41:1378–1387.

Bresciani M, Giardino C, Longhi D, Pinardi M, Bartoli M, Vascellari M. Imaging spectrometry of productive inland waters. Application to the lakes of Mantua. Ita J of Rem Sens 2009;41:147–156.

Bricaud A, Morel A, André JM. Spatial/temporal variability of algal biomass and potential productivity in the Mauritanian upwelling zone, as estimated from CZCS data. Adv Space Res 1987;7:53–62.

Bukata RP, Jerome JH, Kondratyev KY, Pozdnyakov DV. Satellite monitoring of optically-active components of inland waters: an essential input to regional climate change impact studies. J Great Lakes Res 1991;17:470–478.

Buraschi E, Buzzi F, Garibaldi L, Legnani E, Morabito G, Oggioni A, Pozzi S, Salmaso N, Tartari G. Protocollo di campionamento di macrofite acquatiche in ambiente lacustre. Protocollo per il campionamento di fitoplancton in ambiente lacustre. Metodi biologici per le acque. Parte 1. 2008; APAT (Italian Agency for the Environmental Protection).

Carvalho L, Solimini A, Phillips G, van den Berg M, Pietiläinen O-P, Lyche Solheim A, Poikane S, Mischke U. Chlorophyll reference conditions for European intercalibration lake types. Aquatic Ecology 2008;42:203–211.

Cavalli RM, Laneve G, Fusilli L, Pignatti S, Santini F. Remote sensing water observation for supporting Lake Victoria weed management . J Environ Manage 2009;90:2199–211.

Chen Q, Zhang Y, Ekroos A, Hallikainen M. The role of remote sensing technology in the EU Water Frame Work Directive (WFD). Env Sci & Policy 2004;7:267–276.

Chen Q, Zhang Y, Hallikainen M. Water quality monitoring using remote sensing in support of the EU Water Framework Directive (WFD): A case study in the Gulf of Finland. Environ Monit Assess 2007;124:157–166.

Cottingham KL, Rusak JA, Leavitt PR. Increased ecosystem variability and reduced predictability following fertilisation: Evidence from palaeolimnology. Eco Lett 2000;3:340–348.

Dekker AG, Peters SWM, Vos RJ, Rijkeboer M. Remote sensing for inland water quality detection and monitoring: state-of-the-art application in Friesland waters. In: Van Dijk A, Bos MG editors. GIS and Remote Sensing Technology in Land and Water Management. Netherlands: Kluwer Academic Publishers; 2001. p. 17–38.

Directive. Directive 2000/60/EC of the European Parliament and of the council of 23 October 2000 establishing a framework for community action in the field of water policy. Official Journal of the European Communities 2000; L327:1–72.

Doerffer R, Schiller H. The MERIS Case 2 water algorithm. Int J of Rem Sens 2007;28:517–535.

Doerffer R, Schiller H. MERIS Regional Coastal and Lake Case 2 Water Project Atmospheric Correction ATBD, v. 1.0. May 2008a.

Doerffer R, Schiller H. MERIS Lake Water Algorithm for BEAM ATBD, v. 1.0. May 2008b.

Eckmann R, Gerdeaux D, Müller R, Rösch R. Re-oligotrophication and whitefish fisheries management – a workshop summary. Adv in Limn 2007;60:353–360.

Finger D, Bossard P, Schmid M, Jaun L, Müller B, Steiner D, Schäffer E, Zeh M, Wüest A. Effects of alpine hydropower operations on primary production in a downstream lake. Aquat Sci 2007;69:240–256.

Friedrich U, Schallenberg M, Holliger C. Pelagic Bacteria–Particle Interactions and Community-Specific Growth Rates in Four Lakes along a Trophic Gradient. Microb Ecol 1999;37:49–61.

Giardino C, Bresciani M, Villa P, Martinelli A. Application of remote sensing in water resource management: the case study of lake Trasimeno, Italy. Wat Res Manag, 2010;24:3885–3899.

Giardino C, Pepe M, Brivio PA, Ghezzi P, Zilioli E. Detecting chlorophyll, Secchi disk depth and surface temperature in a Subalpine lake using Landsat imagery. Sci of the Tot Env 2001;268:19–29.

Gitelson AA, Schalles JF, Hladik CM. Remote chlorophyll-a retrieval in turbid, productive estuaries: Chesapeake bay case study. Rem Sens Environ 2007;109:464–472.

Gons H J, Auer MT, Effler SW. MERIS satellite chlorophyll mapping of oligotrophic and eutrophic waters in the Laurentian Great Lakes. Rem Sens Environ 2008;112 4098–4106.

Gons HJ, Rijkeboer M, Ruddick K. A chlorophyll-retrieval algorithm for satellite imagery (Medium Resolution Imaging Spectrometer) of inland and coastal waters. J Plank Res 2002;24:947–951.

Gordon HR, Austin RW, Clark DK, Hovis WA, Yentsch CS. Ocean color measurements. Adv Geophys 1985;27:297–333.

Gower J, King S. Validation of chlorophyll fluorescence derived from MERIS on the west coast of Canada. Int J of Rem Sens 2007;28:625–635.

Guilizzoni P, Bonomi G, Galanti G, Ruggiu D. Relationship between sedimentary pigments and primary production: evidence from core analysis of 12 Italian lakes. Hydrobiologia 1983;103:130–106.

Jeppesen E, Jensen JP, Jensen C, Faafeng B, Hessen DO, Søndergaard M, Lauridsen T, Brettum P, Christoffersen K. The impact of nutrient state and lake depth on top-down control in the pelagic zone of lakes: a study of 466 lakes from the temperate zone to the Arctic. Ecosystems 2003;6:313–325.

Kaiblinger C, Anneville O, Tadonleke R, Rimet F, Druart JC, Guillard J, Dokulil MT. Central European water quality indices applied to long-term data from peri-alpine lakes: test and possible improvements. Hydrobiologia 2009;633,67–74.

Koponen S, Kallio K, Pulliainen J, Vepsäläinen J, Pyhälahti T, Hallikainen M. Water quality classification of lakes using 250-m MODIS data MERIS data, IEEE Geosci Remote S 2004;1:287–291.

Kufel L. Uncoupling of chlorophyll and nutrients in lakes–possible reasons, expected consequences Hydrobiologia 2001;443:59–67.

Lindell T, Pierson D, Premazzi G, Zilioli E. Manual for monitoring European lakes using remote sensing techniques. Luxembourg: Office for Official Publications of the European Communities, EUR Report n.18665 EN; 1999.

Manca M, Torretta B, Comoli P, Amsinck S, Jeppesen E. Major changes in trophic dynamics in large, deep sub-alpine Lake Maggiore from 1940s to 2002: a high resolution comparative palaeo-neolimnological study. Freshwater Bio 2007;52:2256–2269.

Nõges P, Poikane S, Kõiv T, Nõges T. Effect of chlorophyll sampling design on water quality assessment in thermally stratified lakes. Hydrobiologia 2010;649:157–170.

Odermatt D, Giardino C, Heege T. Chlorophyll retrieval with MERIS Case-2 Regional in perialpine lakes. Rem Sens Environ 2010;114:607–617.

Personnic S, Domaizon I, Dorigo U, Berdjeb L, Jacquet S. Seasonal and spatial variability of virio-, bacterio- and picophytoplanktonic abundances in three peri-alpine lakes. Hydrobiologia 2009;627:99–116.

Peter A, Kösterc O, Schildknechtc A, von Gunten U. Occurrence of dissolved and particle-bound taste and odor compounds in Swiss lake waters. Water Rese 2009;43:219–2200.

Premazzi G, Dal Miglio A, Cardoso AC, Chiaudani G. Lake management in Italy: the implications of the Water Framework Directive. Lakes & Reser: Rese and Manag 2003;8:41–59.

Pugnetti A, Camatti E, Mangoni O, Morabito G, Oggioni A, Saggiomo V. The phytoplankton production in Italian freshwater and marine ecosystems: state of the art and perspectives. Chemistry and Ecology 2006;22:49–69.

Ruggiu D, Morabito G, Panzani P, Pugnetti A. Trends and relations among basic phytoplankton characteristics in the course of the long-term oligotrophication of Lake Maggiore (Italy). Hydrobiologia 1998;369/370:243–257.

Salmaso N, Mosello R. Limnological research in the deep southern subalpine lakes: synthesis, directions and perspectives. Adva in Ocean and Lim 2010;1:1–47.

Salmaso N, Morabito G, Garibaldi L, Mosello R. Trophic development of the deep lakes south of the Alps: a comparative analysis. Fund. Appl. Limnol. 2007;170:177–196.

Salmaso N. Effects of climatic fluctuations and vertical mixing on the interannual trophic variability of Lake Garda, Italy. Limn Ocean 2005;50:553–565.

Salmaso N. Long-term phytoplankton community changes in a deep subalpine lake: responses to nutrient availability and climatic fluctuations. Freshwater Biol. 2010;55:825–846.

Salmaso N. Seasonal variation in the composition and rate of change of the phytoplankton community in a deep subalpine lake (Lake Garda, Northern Italy). An application of nonmetric multidimensional scaling and cluster analysis. Hydrobiologia 1996;337:49–68.

Salmaso N, Morabito G, Buzzi F, Garibaldi L, Simona M, Mosello R. Phytoplankton as an indicator of the water quality of the deep lakes south of the Alps. Hydrobiologia 2006;563:167–187.

Santer R, Schmechtig C. Adjacency Effects on Water Surfaces: Primary Scattering Approximation and Sensitivity Study. App Opt 2000;39:361–375.

Santer R, Zagolski F. ICOL - Improve contrast between ocean & land ATBD, Université du Littoral, Wimereux, France. Version 1.1, 6 January 2009.

Søndergaard M, Jeppesen E, Jensen JP, Amsinck SL. Water Framework Directive: ecological classification of Danish lakes. J of App Ecology 2005;42:616–629.

Stich HB, Brinker A. Oligotrophication outweighs effects of global warming in large, deep, stratified lake ecosystems. Global Change Biology 2010;16:877–888.

Stramska M, Stramski D. Effects of non-uniform vertical profile of chlorophyll concentration on remote-sensing reflectance of the ocean. App Opt 2005;44:1735–1747.

Strömbeck N, Pierson D. The effects of variability in the inherent optical properties on estimations of chlorophyll a by remote sensing in Swedish freshwater. Sci Total Environ 2001;268:123–137.

Vollenweider RA, Kerekes J. The loading concept as a basis for controlling eutrophication philosophy and preliminary results of the OECD programme on eutrophication. Prog Wat Rech 1980;12:5–38.

Wolfram G, Argillier C, de Bortoli J, Buzzi F, Dalmiglio A, Dokulil MT, Hoehn E, Marchetto A, Martinez P-J, Morabito G, Reichmann M, Remec-Rekar Š, Riedmüller U, Roury C, Schaumburg J, Schulz L, Urbanič G. Reference conditions and WFD compliant class boundaries for phytoplankton biomass and chlorophyll-a in Alpine lakes. Hydrobiologia 2009;633:45–58.