

Ocean Colour Remote Sensing of Harmful Algal Blooms in the Benguela System

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Abstract. The Benguela, as a highly productive upwelling system, suffers from the occurrence of a variety of harmful algal blooms, most of which are associated with elevated biomass; a feature common to the shelf environment of upwelling systems. Most harmful blooms have in the past been attributed to one or another dinoflagellate species, but more recently harmful impacts have also been ascribed to other groups of phytoplankton, including diatom and autotrophic ciliate species. Typical bloom assemblages, forcing mechanisms and harmful impacts are outlined, and bloom types most amenable to detection with ocean colour radiometry are identified. Inherent and apparent optical properties of these algal assemblage types are described, and a preliminary evaluation is made of the suitability of available ocean colour data and algorithms. The evolution of several bloom events is described using various algorithms applied to ocean colour data from the MERIS sensor, and recommendations are made regarding optimal ocean colour usage for high biomass algal blooms in the coastal zone.

1. Introduction

The upwelling systems of the eastern boundaries of the world's oceans support high biological productivity and a high incidence of harmful algal blooms (HABs) (Kudela *et al.*, 2005). The majority of species contributing

to HABs in upwelling systems constitute regular components of the annual succession of phytoplankton, and their harmful impacts are associated with either their toxic properties or the high biomass such blooms can achieve (Trainer *et al.*, 2010). Negative consequences include contaminated seafood, the mortality of fish and other animals, and habitat or ecosystem degradation, all to the detriment of coastal and fishing communities.

There is a strong seasonality to phytoplankton community structure in the southern Benguela, with larger-celled diatoms and dinoflagellates typically dominating inshore waters during the upwelling-driven summer months, and smaller-celled flagellates more dominant in winter (Barlow *et al.*, 2005). Most HABs in the Benguela upwelling system have been attributed to one or another dinoflagellate species, but harmful impacts have also been ascribed to other groups of phytoplankton, including the raphidophytes, diatoms and the ciliate *Myrionecta rubra* (Trainer *et al.*, 2010). Of the impacts associated with toxigenic phytoplankton, those dinoflagellate species responsible for the shellfish poisoning syndromes Paralytic Shellfish Poisoning (PSP) and Diarrhetic Shellfish Poisoning (DSP) pose the greatest risk to human health in the Benguela region (Pitcher & Calder 2000). As a producer of saxitoxins *Alexandrium catenella* is the primary cause of PSP, whereas several species of *Dinophysis*, through the production of okadaic acid and its derivatives, are the causative agents of DSP (Pitcher *et al.*, 2011; Hubbart *et al.*, 2012). Lesser risks to human health in the Benguela are presented by the production of yessotoxins by the dinoflagellate *Protoceratium reticulatum* (Krock *et al.*, 2008), and by the production of domoic acid by diatoms of the genus *Pseudo-nitzschia* (Fawcett *et al.*, 2007; Hubbart *et al.*, 2012). The risk to seafood safety presented by these species can result in significant losses in harvestable resources and represents a significant threat to the aquaculture industry within the region.

In addition to their accumulation in shellfish, some dinoflagellate toxins, such as those produced by species of *Karlodinium* and *Karenia* can lead directly to mortalities of marine life. In the northern Benguela *Karlodinium veneficum* produces a suite of toxic compounds characterised by haemolytic, ichthyotoxic and cytotoxic properties, and has long been associated with massive fish kills (Copenhagen 1953), whereas in the southern Benguela *Karenia cristata*, is not only responsible for faunal mortalities but may impact human health through eye, nose, throat and skin irritations (Botes *et al.*, 2003). Other fish-killing phytoplankton in the Benguela include the raphidophyte *Heterosigma akashiwo*, although the underlying toxicological mechanisms remain uncertain (Pitcher & Calder 2000).

Events of anoxia, and in some cases associated increases in sulphide concentration, represent one of the more conspicuous and well document-

ed impacts of HABs in the Benguela ecosystem (van de Lingen *et al.*, 2006). Anoxia follows the decay of high biomass dinoflagellate blooms, referred to as red tides. Often dominated by species of *Ceratium* and *Prorocentrum*, these blooms develop during late summer and autumn as seasonal stratification strengthens. Wind relaxation and consequent downwelling force the inshore accumulation of these blooms. Here bloom decay, driven by the exhaustion of nutrients, creates a very high oxygen demand though the entire water column (Pitcher & Probyn 2011). The resulting onset of hypoxia and anoxia may cause major perturbations to the diversity, structure and functioning of the coastal environment as witnessed on occasions by mass mortalities of tons of rock lobster and other marine life (Cockcroft *et al.*, 2000). Other ecosystem impacts may be more subtle and difficult to quantify, and may include altered food web interactions and habitats. The poor food source provided by blooms of the pelagophyte *Aureococcus anophagefferens*, owing either to their small size or to a toxic entity, has impacted food chains in this way, with dramatic reductions in the growth rates of filter feeding bivalves (Probyn *et al.*, 2001). Further impacts associated with high biomass blooms, often dominated by a single species, include unsightly discolouration of the water, which may negatively influence the important economic sectors of tourism and recreation.

Globally, more HABs are recorded now than in the past placing a greater need for resource managers and public health officials to be provided with better tools to monitor and possibly forecast imminent HAB events. The association of HABs in upwelling systems with elevated biomass allows spaceborne surveillance of ocean colour to contribute to such an operational capability through the provision of rapid and spatially broad information relating to the development and progression of HABs (Bernard *et al.*, 2006). Ocean colour may therefore contribute fundamental information for the establishment of early warning systems allowing better assessment of the incidence of blooms, and better planning and management options in different coastal regions.

2. Aims of Ocean Colour Application to Harmful Algal Blooms

Ocean colour radiometry offers routine synoptic data pertaining to the phytoplankton biomass and assemblage type in the upper optical depths, and thus offers systematic observations from the event to decadal time scales. On the event scale ocean colour derived observations in upwelling systems

offer enhanced, even unique, capabilities in several domains: freely-available, near real time synoptic data for operational bloom monitoring; the detection of precursive and formative bloom conditions; an enhanced capacity to determine whether a bloom is of a potentially harmful nature; an accessible facility to monitor bloom growth, movement and decay; and a greater understanding of the bio-physical dynamics underlying bloom formation (Bernard *et al.*, 2006, Pitcher *et al.*, 2008a). Over longer time scales, such observations offer insight into upwelling system functionality, inter- and intra-seasonal variability and perhaps even environmental regime shifts (Kahru *et al.*, 2009, Pitcher & Weeks 2006, Weeks *et al.*, 2006). However it must be realised that ocean colour data, dependent on the gross bio-optical characteristics of surface water constituents, is by nature imprecise by comparison with the species-level *in situ* data typically used to determine HAB assemblage characteristics and potentially harmful impacts.

Ocean colour application to HABs is therefore most valuable when contextualised from an ecosystem-specific perspective. Of particular importance is the need to contextualise observations from an ecosystem understanding of phytoplankton succession; to utilise satellite derived information through time and space to identify and understand the competitive success of a particular phytoplankton species or functional type in a given ecological niche. Conceptual succession schemes (Margalef 1978, Smayda 2002) provide a useful framework for ocean colour application in upwelling systems in ecological space: a key aspect is the distinction between diatom and dinoflagellate dominance, primarily controlled by turbulence and nutrient availability (Margalef 1978). Upwelling systems are physically driven, but with a somewhat stochastic determination of phytoplankton functional type dominance within a given ecological niche (Kudela *et al.*, 2010). Beyond the obvious need to detect the magnitude, extent and transport of blooms, one of the key aims of ocean colour application should thus be to elucidate assemblage structure within an environmentally determined ecological niche. HAB application of ocean colour radiometry therefore necessarily introduces the need for a species or at least phytoplankton functional type identifier from ocean colour: a bloom cannot be identified as potentially harmful without some form of phytoplankton assemblage information, even if indirectly derived or highly probabilistic in nature.

Such identifiers can be as simple as an ecosystem-contextualised threshold for phytoplankton abundance, typically determined using chlorophyll *a* concentration as a biomass proxy: such an approach has been successfully used in the Gulf of Florida for the operational detection of

blooms of the toxic dinoflagellate *Karenia brevis* using commonly available ocean colour products (Tomlinson *et al.*, 2009). The southern Benguela system is highly productive; diatom blooms of up to 20 mg m⁻³ chlorophyll *a* are not uncommon in inshore waters (Barlow 1982, Brown & Hutchings 1987), with diatom-dominated biomass occasionally reaching values of > 50 mg m⁻³ (Mitchell Innes & Walker 1991, Fawcett *et al.*, 2007). However, typically the majority of reported blooms displaying enhanced biomass concentrations of > 30-50 mg m⁻³ are dinoflagellate dominated (Fawcett *et al.*, 2007, Pitcher *et al.*, 2008a,b, Pitcher & Probyn 2011, Pitcher & Weeks 2006), and thus more likely to result in harmful impact. Such enhanced biomass is associated with cellular motility and the ability to regulate position in the water column, resulting in enhanced near-surface aggregation of flagellated cells (Franks 1992, Smayda 1997). The biomass range threshold of 30-50 mg m⁻³ chlorophyll *a* can be used as a probabilistic first-order identifier of bloom type: it is likely that a bloom with biomass above this range is dinoflagellate dominated. However, the accurate determination of chlorophyll *a* concentrations > 20 mg m⁻³ is well outside the scope of commonly available algorithms for the MERIS sensor, both for Case 1 waters (Morel *et al.*, 2007), and Case 2 waters (Schiller and Doerffer 1999, 2005), and the demonstration of new empirical and semi-analytical algorithms designed for such high biomass application is a primary aim of this study.

The provision of some form of direct assemblage description from ocean colour measurements offers significant advantages over descriptors derived indirectly from abundance estimates; namely an ability to distinguish a greater variety of assemblage types, with greater confidence and in a wider variety of bloom conditions. There are several algorithm types that could be used to achieve this (IOCCG 2006): inherent optical property based reflectance inversion algorithms will be focused on here as amongst the least constrained and therefore potentially most able to provide output pertinent to a variety of bloom types (*ibid.*). Such algorithms are necessarily dependent upon an ecosystem-specific characterisation and parameterisation of the inherent optical properties (IOPs) of the phytoplankton assemblage, and the effects of these upon the water-leaving radiance/reflectance. Use of such algorithms requires some consideration of the major cellular and assemblage-specific parameters causal to bio-optical variability; and quantitative consideration of the ability to detect the effects of such phenomena in the water-leaving radiance signal across a range of phytoplankton biomass – a complex issue, and one as yet unresolved (e.g. Alvain *et al.*, 2012, Dupuoy *et al.*, 2011). Nevertheless key assemblage characteristics potentially identifiable through determination of

phytoplankton IOPs from the water-leaving radiance spectral signal can be considered as:

1. Average assemblage size. Size, as determined through mean assemblage metrics such as the effective diameter (Bernard *et al.*, 2006), is arguably one of the most important gross determinants of the optical properties of the algal assemblage (Morel & Bricaud 1981, Roy *et al.*, 2011, Zhou *et al.*, 2012). In addition, size-based ocean colour algorithms have shown effective application in both the coastal (Ciotti & Bricaud 2006) and open ocean (Kostadinov *et al.*, 2010). Detection of changing assemblage size is of considerable potential value in HAB applications: bloom onset is often characterised by changes in the gross size of the algal assemblage, and arguably blooms dominated by large dinoflagellate cells are most commonly associated with harmful impact in the southern Benguela (Pitcher & Calder 2000).

2. Gross changes in intra-cellular accessory pigments. Whilst the effects of pigment changes on cellular absorption are significant and well understood (Johnsen *et al.*, 1994, Lohrenz *et al.*, 2003), there is considerable debate as to how detectable such effects are on the water-leaving reflectance across different biomass and other optical regimes (Dierssen *et al.*, 2006, Jackson *et al.*, 2011, Alvain *et al.*, 2012). One important determinant in detecting absorption-related effects, in a reflectance signal, is phytoplankton biomass. There is little doubt that, while a detailed sensitivity study still must be conducted, such effects are much more visible, and perhaps only detectable, at considerable biomass concentration (Dierssen *et al.*, 2006). In the southern Benguela, it is highly unlikely that it is possible to distinguish even high biomass blooms dominated by diatoms or dinoflagellates based only on pigment-related absorption effects, given that fucoxanthin and peridinin (as the primary respective accessory pigments) are spectrally similar (Bidigare *et al.*, 1990). However, the photosynthetic ciliate *Myrionecta rubra*, a large-celled, bloom-forming species displaying considerable phycoerythrin absorption (Kyewalyanga 2002), is potentially detectable based on absorption effects on the water leaving reflectance, and such an approach is investigated during this study.

3. Ultrastructure. The presence of intracellular organelles of appreciably different refractive index to the surrounding cell, e.g. gas vacuoles in cyanobacterial species, or calcium carbonate platelets associated with coccolithophores, can have considerable impact on cellular scattering properties (Volten *et al.*, 1998, Whitmire *et al.*, 2010) and thus ocean colour

(Bracher *et al.*, 2009, McKinna *et al.*, 2011, . Blooms dominated by vacuolate species are not of ecological consequence in the southern Benguela and will not be considered further here. Whilst coccolithophore blooms occur in the Benguela, and can be discriminated using ocean colour radiometry (Henderiks *et al.*, 2012, Weeks *et al.*, 2004), such blooms are not associated with harmful impacts and are not considered further here.

4. Fluorescence line height and quantum yield. The MERIS and MODIS sensors have a set of bands, centred at the chlorophyll *a* fluorescence emission peak of ± 683 nm, to measure the sun induced fluorescence of the phytoplankton assemblage e.g. Gower *et al.*, 1999. This signal has a complex dependency on phytoplankton biomass, taxonomy and physiological status (Behrenfeld *et al.*, 2009, MacIntyre *et al.*, 2010). There is evidence indicating that diatoms may exhibit elevated fluorescence quantum yields due to photo-machinery able to rapidly respond to short term variability in photo-acclimation in a high-mixing environment (Lavaud *et al.*, 2002, Nymark *et al.*, 2009). There is thus potential to use fluorescence quantum yield to discriminate between diatom- and dinoflagellate-dominated blooms, of considerable value to HAB application in the Benguela, and such an approach is explored further below.

An example of the large differences in the spectral reflectance, with associated and causal differences in the measured phytoplankton absorption (as per Bernard *et al.*, 2006) and modelled backscattering coefficients (as per Bernard *et al.*, 2009) - as the principle IOPs determining ocean colour - can be seen in Figure 1. Data from a diverse range of bloom types, dominated by a range of assemblages, are shown: a mono-specific bloom (Chl *a* = 12.8 mg m⁻³) of the very small-celled pelagophyte *Aureococcus anophagefferens* (Probyn *et al.*, 2010); a very high biomass bloom (Chl *a* = 309.0 mg m⁻³) of the PSP-toxic, large-celled, chain-forming dinoflagellate *Alexandrium catenella* (Bernard *et al.*, 2009); a high biomass bloom (Chl *a* = 172.5 mg m⁻³) of the relatively small celled dinoflagellate *Prorocentrum triestinum* (*ibid.*); a moderate biomass bloom (Chl *a* = 18.0 mg m⁻³) of the autotrophic, large celled, phycoerythrin containing ciliate *M. rubra* (unpublished); and a moderate biomass bloom (Chl *a* = 20.7 mg m⁻³) dominated by a variety of diatom species.

The *Aureococcus* bloom (with cell counts of $\pm 1.5 \times 10^9$ cells l⁻¹) is notable for very high reflectance values (red spectra, Figure 1A), caused by the enhanced backscattering resulting from the $\pm 2 \mu\text{m}$ effective diameter cell size (Figure 1C), which also results in the notably high Chl-specific phytoplankton absorption values (Figure 1D) (Quirantes & Bernard 2006). This

can be contrasted with the *Alexandrium* bloom, where the large effective diameter of $\pm 30 \mu\text{m}$ results in Chl *a* values > 20 times larger than the *Aureococcus* bloom, despite cell counts ($\pm 9.8 \times 10^6 \text{ cells l}^{-1}$) \pm two orders of magnitude smaller. The *Alexandrium* bloom is notable for attributes associated with large cells at high biomass: the high magnitude, highly-packaged (i.e. spectrally relatively flat) phytoplankton absorption values (blue spectra, Figure 1D); the relatively low backscattering (Figure 1C) for such elevated biomass, and the relatively low reflectance values dominated by peaks at $\pm 570 \text{ nm}$ and 709 nm (Figure 1A).

Again, there is interesting contrast with the *Prorocentrum* bloom, of effective diameter $\pm 13 \mu\text{m}$ and cell counts of $\pm 1.2 \times 10^8 \text{ cells l}^{-1}$. The smaller cell size relative to the *Alexandrium* assemblage results in reflectance of similar shape (pink spectra, Figure 1A) but enhanced magnitude, an obvious result of the much greater phytoplankton backscattering (Figure 1C) associated with the smaller cell size (Bernard *et al.*, 2009). The *Myrionecta* bloom displays reflectance features associated with considerable phycoerythrin absorption (the bifurcated peaks at $\pm 530 \text{ nm}$ and 600 nm , green spectra, Figure 1A), and the effects of slightly lower biomass relative to the two dinoflagellate assemblages: greater reflectance at blue wavelengths, and a longer wavelength peak $< 700 \text{ nm}$ where Chl *a* fluorescence is still observable (Figure 1A).

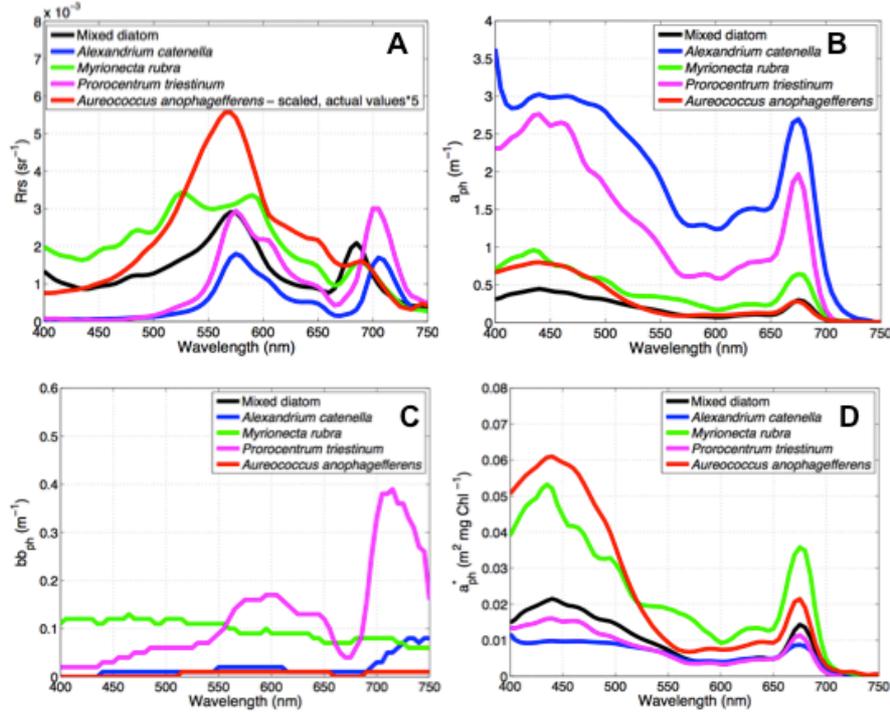


Fig. 1. Optical properties of five diverse blooms occurring between 1998 and 2005 from various locations in the southern Benguela: remotely-sensed reflectance R_{rs} (A), phytoplankton absorption coefficients a_{ph} (B), phytoplankton backscattering coefficients b_{bph} (C), and Chl-specific phytoplankton absorption coefficients a^*_{ph} (D).

Such data demonstrate the potential utility of using *in situ* bio-optical and ocean colour-analogous data to distinguish between bloom types. However, there is still a considerable challenge to utilising such data from space-borne ocean colour sensors. The retinue of methodological challenges to ocean colour application in the coastal environment are compounded by HAB application: atmospheric correction in turbid waters and an aerosol-laden atmosphere; distinction of bio-optical constituents in optically complex waters; and adjacency effects from land. The phytoplankton dominated waters of the Southern Benguela can in the large part be conventionally classified as Case 1 (Morel & Prieur 1977), excluding the typically short-lived case of freshly upwelled waters where there are enhanced concentrations of small inorganic particulate through the water column

(Kudela *et al.*, 2006). However, the considerable phytoplankton biomass observed under bloom conditions in the Benguela provide the same functional obstacles to atmospheric correction algorithms as sediment-influenced Case 2 waters: enhanced turbidity and resulting non-zero signal in the near-infrared, thereby compromising atmospheric correction algorithms assuming null water-leaving radiance signal in NIR bands (Gordon & Wang 1994, Wang 2006). Past use in the Benguela has shown that the Bright Pixel Atmospheric Correction (BPAC) algorithm (Moore & Lavelander 2011), with an ability to adjust for non-zero NIR reflectance, performs relatively well on data from the MERIS sensor in high biomass Benguela waters. In addition, the typical high biomass reflectance signals of interest here contain a strong signal, if not a distinct peak, in the 700 – 730 nm spectral range (Matthews *et al.*, 2012, Pitcher *et al.*, 2008a): the only recent ocean colour mission carrying an appropriately located spectral band, at 709 nm, is the MERIS sensor (and forthcoming OLCI sensor on the Sentinel 3 series). Reduced resolution (± 1 km ground resolution) data from the MERIS sensor will therefore be exclusively focused on here; in addition to optimal band placement and routine availability of suitably derived water-leaving reflectance data, the sensor has excellent sensitivity with a large signal:noise ratio, and suitably high revisit time with global coverage every two days. This study seeks to demonstrate the utility of products derived from these ocean colour data to identify bloom types based on cell size and intracellular pigment characteristics, and show ocean colour application to bloom observation at the event scale.

3. Methods: Bio-optical data and algorithms

St Helena Bay in the Southern Benguela is focussed on due to its high biomass levels and the frequent occurrence of Harmful Algal Blooms within this region. A retentive circulation, linked with the upwelling dynamics of the bay, plays an important role in the transport, concentration and dissipation of blooms (Pitcher & Calder, 2000). As such, St Helena Bay has been the site of numerous *in situ* data collection campaigns during the last decade. In addition to the satellite data and algorithm output presented here, *in situ* radiometry from a Satlantic Hyperspectral Tethered Surface Radiometer Buoy (H-TSRB) and *in situ*, fluorometric chlorophyll values are provided for comparison. Absorption measurements were made from filtered samples using a Shimadzu UV-2501 spectrophotometer. Identification of species and estimates of effective diameter from particle size distributions were made through microscopy and use of a Beckman Coulter Counter re-

spectively. Sampling was generally conducted from Lamberts Bay at stations around -32.08 N and 18.26 E, with additional sampling where bloom patches were evident. Detailed laboratory and processing methodology can be found in Bernard *et al.*, 2009.

Two algorithms are used in the study. The Maximum Peak Height (MPH) algorithm (Matthews *et al.*, 2012) is an empirical top-of-atmosphere algorithm designed for MERIS application in high biomass waters, returning only Chl *a* concentrations. The Equivalent Algal Population (EAP) algorithm (variant of Bernard *et al.*, 2005) is a semi-analytical reflectance algorithm, based upon inversion of IOPs from modelled algal populations, which gives a more comprehensive suite of returns.

The EAP algorithm can be summarised as follows. Inputs are MERIS atmospherically corrected multi-spectral normalised water leaving reflectance, on a pixel-by-pixel basis. The model uses five solvable unknowns: chlorophyll *a* concentration (Chl *a*, mg m⁻³), algal effective diameter (D_{eff} , μm), the relative concentration of two representative algal groups (diatoms/dinoflagellates and nanoflagellates/chlorophytes), combined gelbstoff and detrital absorption (e.g. $a_{\text{gd}}(400)$, m⁻¹), and small particle backscattering (e.g. $b_{\text{bs}}(550)$, m⁻¹). Gelbstoff/detrital absorption and small particle backscattering employ constant spectral shapes and variable magnitude (Bernard *et al.*, 2005).

The EAP algorithm uses an unconstrained non-linear minimisation (Nelder & Mead, 1965), with constant initial values except for chlorophyll, for which an initial estimate is provided via an empirical, band-ratio switching algorithm on a per pixel basis. This switching algorithm extends the maximum band ratio approach used in the standard MERIS algal 1 product, by adding an additional ratio between 665 nm and 709 nm bands. The EAP algorithm and all subroutines are coded in Matlab R14. As the underlying model specifically does not account for sun-induced natural fluorescence, the convergence weighting for the Nelder-Mead solution is set to negligible values between 665 and 715 nm – the spectral region affected by natural algal fluorescence. The algorithm thus does not seek to match spectral reflectance values at these fluorescence wavelengths, and in effect offers a means of discriminating fluorescence effects, as have earlier models of a similar nature (Roesler & Boss 2003). This allows the derivation of algal fluorescence quantum yield as an additional algorithm product, utilising integrated fluorescence (calculated from fitting a Gaussian distribution to modelled R_{rs} subtracted from measured R_{rs} at fluorescence wavebands), algal absorption as returned by the algorithm, and incident scalar irradiance calculated independently (Gregg & Carder 1990).

4. Case Studies: Application to Bloom Types

4.1 Transitions between dominant species detected through changes in effective diameter using the EAP algorithm

The EAP algorithm approach allows for determination of a wide range of chlorophyll concentrations and an “effective diameter” relating to the particle size distribution of the algal assemblage. In the case of mono-specific blooms this will equate approximately to the average cell size of the dominant species. Imagery from 30 March and 5 April 2005 processed with the EAP algorithm allows for detection of the elevated biomass levels within the greater St Helena Bay region through this period (*in situ* values between 172.5 mg m^{-3} and 39.6 mg m^{-3}). In addition, a significant change in effective diameter of the assemblage is detected (Figure 2E and 2F). Concurrent *in situ* measurements (microscopy and coulter counter derived size distributions) suggest that this change in size represents a change in dominance of this bloom from the small-celled dinoflagellate *Prorocentrum triestinum* to the large-celled dinoflagellate *Ceratium furca*.

Comparisons between *in situ* (Satlantic H-TSRB) and satellite (MERIS) derived radiometry (Figures 2A and 2B) show that the satellite captures both the magnitude and shape of the spectra in each case. The significant signal in the NIR bands seen here triggers the use of the bright pixel atmospheric correction (BPAC) over large portions of the images, particularly in areas associated with high chlorophyll concentrations.

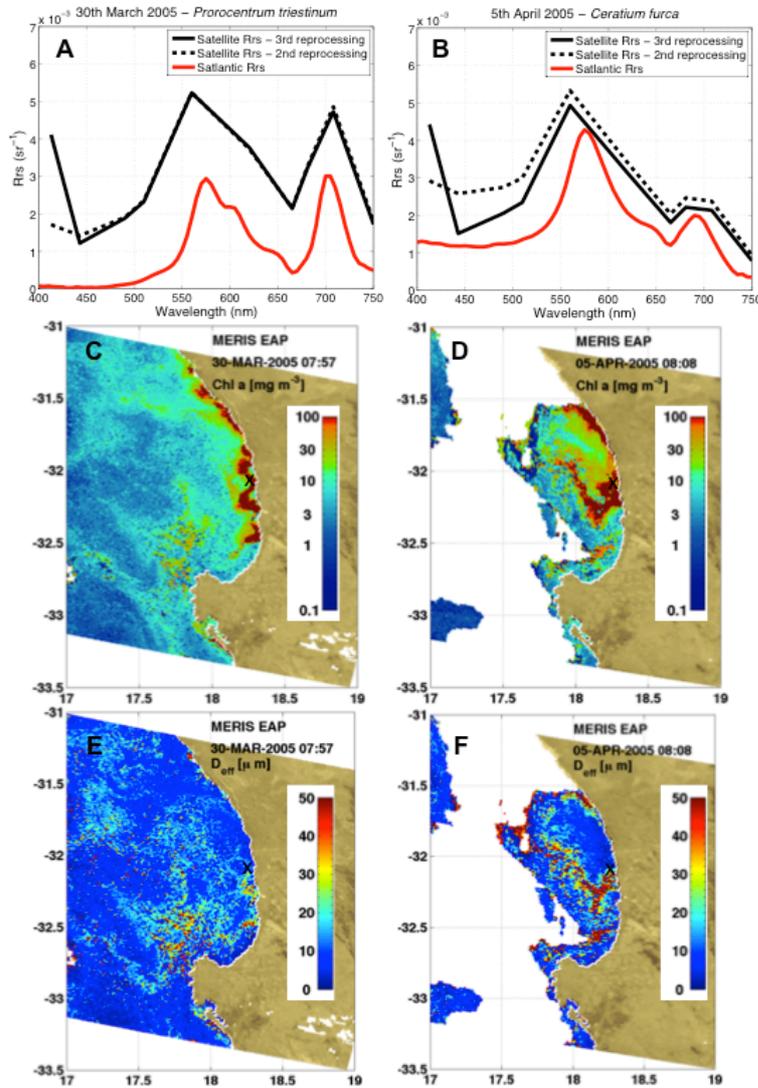


Fig. 2. MERIS RR radiometric match-ups and algorithm outputs from the EAP semi-analytical algorithm, demonstrating the effectiveness of this approach for detecting high biomass (C and D) and distinguishing dominance by small (*Prorocentrum triestinum*) and large (*Ceratum furca*) dinoflagellates (E and F respectively) using the effective diameter product. Radiometric match ups show consistent Bright Pixel Atmospheric Correction performance across 2nd and 3rd MERIS reprocessing, though some overestimation is seen (A and B).

4.2 Use of fluorescence and an assessment of pigment detection using hyperspectral and multispectral radiometry using the EAP algorithm and spectral flags.

Whilst detection of cell size provides a useful ecosystem parameter and potential indicator of taxonomic shifts, the assessment of HAB impact often requires species identification, particularly where toxic effects are being considered. As previously mentioned, a shift from diatom to dinoflagellate dominance forms a key part of many conceptual frameworks for phytoplankton succession in upwelling systems (Margalef 1978). An ability to detect this shift using satellite data may help provide early prediction for the onset of HABs, as well as for comparison to oceanographic data to validate these succession schemes.

Though differentiating between diatoms and dinoflagellates is useful, an ability to identify particular species would be optimal in cases where toxicity is suspected. The existence of accessory pigments with specific absorption characteristics provides a way that individual species can be identified from radiometric data. However the extent to which these spectral characteristics can be exploited is dependent on both the level of biomass and the specifications of instruments used, particularly in the case of satellite derived radiometry. Figure 3A shows match-up *in situ* and MERIS reflectance associated with a relatively low biomass ($\text{Chl } a = 3.5 \text{ mg m}^{-3}$) bloom in the St Helena Bay area containing *Myrionecta rubra*. Figure 3B displays a variety of *in situ* R_{rs} spectra from a three week period during the same bloom event; spectra chosen to show naturally occurring admixtures of diatoms and *M. rubra* at a range of biomass. Peaks related to phycoerythrin absorption can be clearly seen in the data from the *in situ* Satlantic H-TSRB between 500 nm and 620 nm. However, from satellite it is difficult to fully resolve these peaks due to a paucity of bands in this region of the spectra (Figure 3A), leading to difficulties during the inversion procedure. In lower biomass cases (Figure 3C), the generation of the effective diameter fails to retrieve a reasonable estimate (around 30 μm for monospecific blooms) (Figure 3D). In high biomass cases however, such as for the bloom shown in Figure 3E, where more reasonable estimate of effective diameter can be retrieved (Figure 3F), a ratio between bands 620 nm and 550 nm can be used to further identify the presence of *Myrionecta rubra* (Figure 3H) from a background of mixed diatom blooms. These examples show the complexities of detecting accessory pigment-based optical variability with multi-spectral ocean colour data, in particular the very strong effects of biomass on the useable reflectance signal.

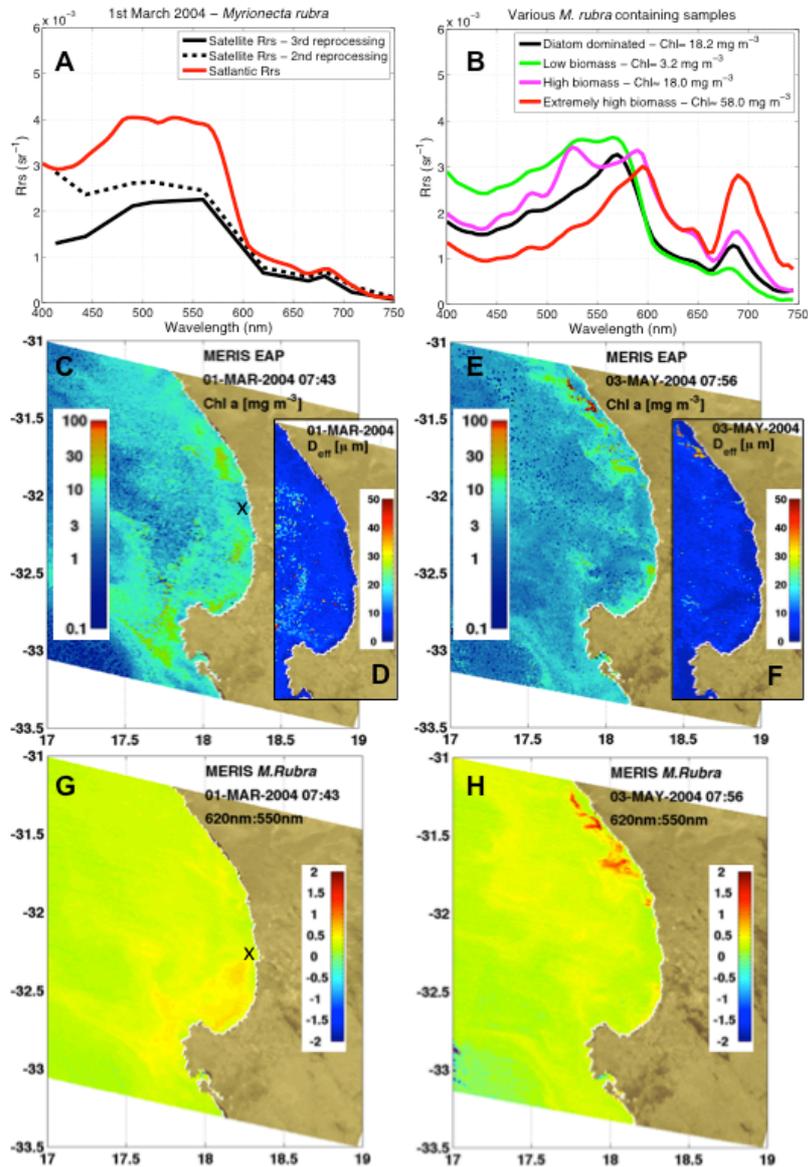


Fig. 3. MERIS RR radiometric match-ups (A) for 1st March, and a variety of R_{rs} spectra from a three week period from the same bloom event (B). Algorithm outputs from the EAP semi-analytical algorithm: Chl a concentration (C) and effective diameter (D) from 1st March 2004; Chl a concentration (E) and effective diameter (F) from 3rd May 2004; and 620:550nm R_{rs} ratio from 1st March (G) and 3rd May (H).

4.3 Tracking the development of a *Ceratium balechii* bloom and consequent anoxia using the MPH algorithm.

In the autumn of 2009 the development and ultimate decay of an exceptional bloom of the dinoflagellate *Ceratium balechii* lead to an anoxia-induced mass mortality of macrofauna in St Helena Bay (Pitcher and Probyn 2011). Initial build-up of the bloom was evident in mid-February from chlorophyll concentrations derived by application of the MPH algorithm to MERIS data (Figure 4). The bloom was first observed to the north of St Helena Bay in the nearshore region of the southern Namaqua shelf and was on occasions shown to extend in a narrow band over a distance of 100 km (Figure 4; 14 March 2009). Under diminished upwelling activity during late summer and early autumn inshore counter currents resulted in the southward progression of the bloom and its entrainment into the Bay (Figure 4; 11 April 2009). In early May the bloom was shown to have accumulated in the shallow, southern reaches of the Bay (Figure 4; 1 May 2009). Degradation of the bloom occurred in these shallow waters where subthermocline nutrients, necessary for bloom maintenance, are inaccessible. The exceptional organic loading of the system as afforded by the bloom resulted in anoxia through the entire water column. Large fish and lobster mortalities were consequently observed off the Berg River estuary and adjacent beaches on the 5 May 2009. Termination of the boom was associated with persistent downwelling conditions resulting in advection of the bloom from St Helena Bay shortly after the anoxic event (Figure 4; 4 May 2009). Surveillance of these blooms as depicted in Figure 4 enables continual assessment by fisheries and coastal managers of the risk posed by these phenomena to marine resources and the coastal environment.

The case study demonstrates the ability of even relatively simple algorithms, such as the MPH, to effectively track very high biomass blooms. It should further be noted that several of the images ($\pm 40\%$) suffered from high sun-glint, which would prohibit use of EAP-type algorithms that use the full spectrum of water-leaving reflectance data after aerosol and Rayleigh atmospheric correction. The MPH is less susceptible to glint effects, as it uses only a small set of green to NIR wavebands and does not employ an aerosol correction (Matthews *et al.*, 2012): this allows increased frequency of image utility on the event scale, a significant advantage for operational application.

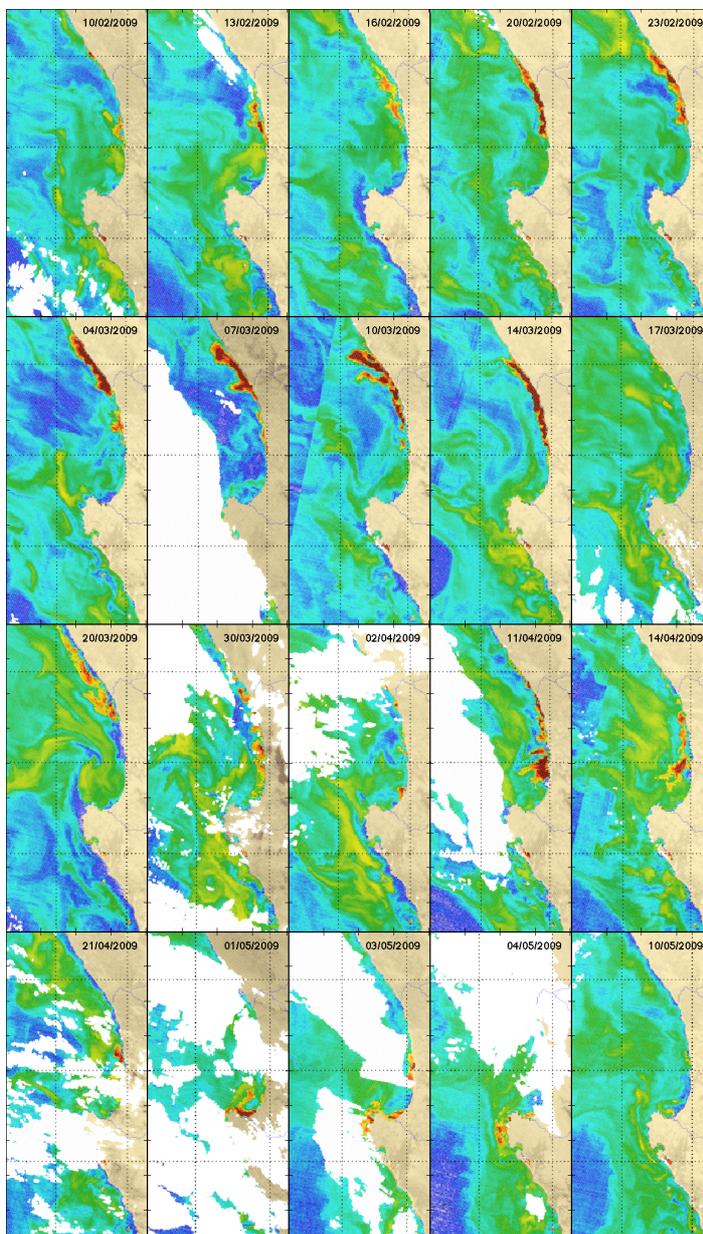


Fig. 4. A montage of MPH derived Chl *a* output showing the development and transport of a bloom of *C. balechii* into St Helena Bay. Consequent decay of the bloom in the nearshore environment led to conditions of anoxia and the mortality of marine life (Pitcher & Probyn 2011).

5. Summary and Outlook

Ocean colour products show great potential for enhancing our ability to both detect and quantify Harmful Algal Blooms, in particular for the very high biomass blooms typically encountered in the Benguela. It should be stressed that many toxic algal species can have harmful effects at low biomass (e.g. McGillicuddy *et al.*, 2005); the algorithms presented here, and arguably perhaps even ocean colour data in general, may not be effective for such bloom types. Combined with an understanding of ecosystem dynamics, satellite derived products can be used to detect elevated biomass and indicate potential for occurrence of toxic species at appropriate temporal and spatial scales. This work highlights the importance of band selection on satellite-mounted sensors, particularly when wishing to exploit pigment-related effects and fluorescence-based products. It is apparent that accurate atmospheric correction in turbid waters must be a first priority to fully utilise these spectrally dependent algorithms. Alternatively, top of atmosphere approaches such as the MPH algorithm used here can circumvent this requirement. The frameworks provided in this study should be adaptable for use in other systems where Harmful Algal Blooms occur. For systems where species have broadly similar inherent optical properties (IOPs) to those in the Benguela, a semi-analytical inversion approach such as that used in the EAP algorithm, demonstrated with the case study in Section 4.1, should be able to provide improved chlorophyll estimates and an indication of the effective diameter of the assemblage. Where mono-specific blooms are prevalent, this parameter is most useful in detecting changes in dominance. This method may prove less successful where other constituents dominate the optical signal, e.g. in severely sediment loaded waters. In cases where species are present which have obscure spectral signatures, resulting from accessory pigments or ultrastructure, new IOPs would need defining, however empirical, spectral ratio techniques such as those used here to detect *Myrionecta rubra* in Section 4.2, could also be applied, given sufficient biomass.

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