Water quality monitoring: a combined approach to investigate gradients of change in the Great Barrier Reef, Australia

J. Udy a,*, M. Gall b, Ben Longstaff c, Kate Moore a, Chris Roelfsema a, D.R. Spooner d, Simon Albert a

a University of Queensland, Qld. 4072, Australia
b NIWA Christchurch, P.O. Box 8602, New Zealand
c Environmental Protection Agency, Qld. 4068, Australia
d NIWA Australia, Brisbane, P.O. Box 359, Qld. 4051, Australia

Abstract

For the managers of a region as large as the Great Barrier Reef, it is a challenge to develop a cost effective monitoring program, with appropriate temporal and spatial resolution to detect changes in water quality. The current study compares water quality data (phytoplankton abundance and water clarity) from remote sensing with field sampling (continuous underway profiles of water quality and fixed site sampling) at different spatial scales in the Great Barrier Reef north of Mackay (20°S). Five transects (20–30 km long) were conducted from clean oceanic water to the turbid waters adjacent to the mainland. The different data sources demonstrated high correlations when compared on a similar spatial scale (18 fixed sites). However, each data source also contributed unique information that could not be obtained by the other techniques. A combination of remote sensing, underway sampling and fixed stations will deliver the best spatial and temporal monitoring of water quality in the Great Barrier Reef.

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1. Introduction

The Great Barrier Reef (GBR) World Heritage Area in Australia, extends over 2000 km along Queensland’s coastline from Lady Elliot Island in the south to the tip of Cape York Peninsula. Within this large expanse of water the outer reef, central lagoon and coastal fringe are linked, but possess different hydrological (Wolanski and Spagnol, 2000; Fabricius and Wolanski, 2000; Brinkman et al., 2002) and water quality characteristics (Brodie et al., 1994; Furnas and Brodie, 1996; Brodie and Furnas, 1996). The mainland coastal fringe (extending approximately 15 km from the coast) is subjected to episodic delivery of sediments and nutrients, with terrestrially derived sediment transport and resuspension processes having their greatest impact in this section (Haynes and Michalek-Wagner, 2000; Neil et al., 2002). Although the reef ecosystem in close proximity to the coastline has evolved in association with episodic flood plumes, anthropogenic changes to land cover have led to higher sediment and nutrient loads being delivered from the GBR catchments than identified in historical analysis (Neil et al., 2002).

The increase in sediment, nutrient and toxicant loads delivered by coastal rivers, particularly during flood events, may pose a threat to the health of GBR ecosystems (Haynes and Michalek-Wagner, 2000; Brodie et al., 2001). Currently three to five times more sediment enters the GBR near-shore environments, than is likely to have occurred when the catchment was under natural
vegetation cover (Neil et al., 2002). Hence, coastal and nearshore ecosystems adjacent to cleared catchments are most at risk because terrestrial material delivered by coastal rivers to the GBR is generally restricted to the coastal waters less than 15 km off the coast (Haynes and Morris, 2003).

In recognition of these environmental concerns the Reef Water Quality Protection Plan (2003) targets non-point source pollutants (primarily sediments and nutrients) that enter waterways in the GBR catchments, eventually impacting on the GBR. It also highlights the need to monitor changes in water quality and various indicators of ecosystem health to determine if the management actions on land are having a noticeable impact on the ecosystems and water quality of the GBR.

Any assessment of water quality or ecosystem health is a compromise between the practicality of measuring the various indicators of interest with the requirements to sample spatial and temporal dynamics at appropriate scales. Hence, a monitoring program for the GBR needs to include a variety of sampling techniques on both small (10’s m) and large (100’s km) spatial scales over short (days) and long (years) periods. These may include: water column sampling at fixed stations from ship-board oceanographic sampling, mooring deployments, underway sampling (provides continuous water quality data along transects) and/or remote observations from satellite sensors, which provide the greatest spatial coverage at a specific point in time. Each technique provides a different insight into the spatial and temporal dynamics of environmental parameters.

The use of remote sensing of ocean colour to determine water turbidity and phytoplankton abundance (chlorophyll $a$) based on radiative transfer equations provides the best spatial coverage possible and is the easiest way to observe water quality patterns in the marine environment (Gordon and Wang, 1994; Carder et al., 2003). However, to understand the absolute concentrations represented by the remotely sensed images or to investigate small scale variability (vertical and horizontal) it is necessary to undertake field sampling. Hence, the primary objective in the current study was to use multiple sampling techniques to characterise the spatial variability of two important coastal water quality parameters: water clarity (estimated by suspended sediments concentration and light attenuation) and phytoplankton abundance (estimated by chlorophyll $a$ concentration) in the Whitsundays region of the GBR.

The current study combined three sampling strategies: (1) detailed measurement at fixed positions, (2) underway sampling of the two dimensional (surface to the bottom) characteristics along five transects perpendicular to the coastline and (3) use of remotely sensed data to provide a two dimensional image of surface water quality parameters. As this near shore and coastal region of the GBR has the largest gradients in water quality (from clear oceanic waters (case 1) to turbid near-shore waters (case 2), Mobley, 1994), it provides an excellent environment to compare and contrast currently available techniques for measuring water quality and discuss how a combined sampling approach can be used to provide a better understanding of coastal water quality and ecosystem dynamics than any one technique used in isolation. Previous studies have compared field data with various satellite borne sensors (Gower, 2004), while others have compared continuous underway sampling using sensors with spot sampling of water quality using standard laboratory extraction methods (Pinto et al., 2001). However, this is the first study of which we are aware of, where multiple sampling techniques for water clarity and chlorophyll $a$ have been used to provide a continuous three dimensional image of water quality on the GBR.

2. Methods

2.1. Study site description

The current study was conducted in the Whitsundays region ($20^\circ–21^\circ$ S, $149^\circ$ E) in the central Great Barrier Reef Lagoon along the Queensland coast, Australia (Fig. 1). The region is characterised by a mainland coastline consisting of many bays and inlets with predominantly muddy foreshores and numerous continental islands with sandy or rocky foreshores including fringing reefs, seagrass beds and mangrove habitat. The main industries in the area are tourism and agriculture. Tourism is based around the tropical reefs and

![Fig. 1. Whitsunday region of the Queensland Coast showing the 5 North East–South West transects conducted using the Biofish and 35 fixed station locations.](image-url)
islands where water clarity is high. Major agricultural activities are sugar cane cultivation (in the river catchments on the coastal floodplains) and cattle grazing (in the hinterland). The two rivers that influence this section of coastline are the Pioneer River (catchment size 1584 km²) (21°7′ S and 149°12′ E), just south of the study region and the Proserpine River (catchment size 2535 km²) (20°30′ S, 148°45′ E), that enters the coast in the middle of the study region.

2.2. Sampling methods

All field sampling was conducted between the 12th and 20th of March 2004. A combination of traditional fixed station sampling techniques were used in combination with continuous underway sampling to provide a range of in situ data that measured water clarity and phytoplankton abundance using different analyses and at different spatial scales. More detailed analysis of water quality variables was conducted at the fixed stations (water samples collected and light quantity and quality analysed), while the continuous underway sampling provided a two dimensional sampling of the water column from 1 m below the surface, down to a maximum depth of 40 m. To provide a more complete spatial coverage of the surface waters, freely downloadable remotely sensed data products from the MODIS (Moderate Resolution Imaging Spectroradiometer) sensor were also used.

During the sampling period, wind was predominantly from the southeast, ranging between 15 and 20 knots most of the time, with occasional readings above or below this range. This wind generated choppy conditions, with wave heights between 1.5 and 3.0 m. Conditions were often overcast with 10–80% cloud cover being recorded during fixed station sampling.

2.2.1. Transect locations

Sampling took place along five transect lines perpendicular to the coastline and the major axis of tidal flow. The transects were spread from 10 km north of Airlie Beach south to Mackay Harbour (21.039 S). They ranged in length from 13 NM to 22 NM (Fig. 1, Table 1). All transects began near a hydrological boundary formed by one of the islands in the Whitsunday group and ended in near shore water adjacent to the mainland, as close as possible to a headland or when the water depth decreased to less than 5 m.

2.2.2. Fixed station sampling

Thirty five fixed stations were sampled during the sampling period (Fig. 1). Sixteen of these stations were at the beginning and end of each transect with samples being collected to measure dissolved nutrients (NO₃, NH₄ and FRP), filterable chlorophyll a and total suspended solids (TSS) of the surface water as well as depth profiles of chlorophyll a fluorescence and turbidity, using a water quality probe (YSI 6600). Downwelling spectral irradiance (W m⁻²) profiles were also measured at the beginning and end of all transects using a spectrometer (Trios RAMSES) and light attenuation was determined from Secchi disk depth (Preisendorfer, 1986). The remaining 19 stations were located along the transects, with water samples being collected from the water surface using a bucket whilst the vessel was moving at approximately 5 knots. At the mid-transect stations it was only possible to measure surface water levels of dissolved nutrients, filterable chlorophyll a and TSS as well as fluorescence and turbidity, using the YSI probe.

2.2.3. Underway water quality sampling

Underway sampling using a towed body probe (Biofish; ADM-Elektronik, Germany) sampled a total horizontal distance of 200 km (Fig. 1). The water column was sampled from the surface water (top 1–2 m of water column) to within 2 m of the bottom, approximately every 50 m along each transect—equating to 2564 vertical profiles. Vertical resolution was typically 0.25 m, but depended on the speed of descent and ascent of the Bio-
fish. The Biofish was towed at approximately 5 knots, with its vertical depth controlled from the research vessel. Once deployed, the Biofish continually moved between the surface and bottom waters collecting water quality data (temperature, conductivity, dissolved oxygen, chlorophyll a fluorescence and transmittance) and relaying it back to a computer on board the research vessel.

A graphical user interface (GUI) was written in Mat-lab (Mathworks Inc.) to process each transect file including removal of data-spikes and near surface data, when the instrument surfaced, as well as combining a mean station co-ordinate (latitude and longitude) to each vertical up and down cast. GUI processed data was imported into Ocean Data View for visualisation and contour plotting (Ocean Data View, http://www.awi-bremerhaven.de/GEO/ODV). Further statistical processing was undertaken using Statistica 6 (Statsoft, Tulsa). Hysteresis was evident in some parameter datasets due to the different up and down trajectories. Although contour grid means smoothed these perturbations, other processing required smoothing using running averages.

2.2.4. Remote sensing sampling

MODIS sensors on two satellites (Terra and Aqua) provide complete global coverage every one to two days, acquiring data in 36 spectral bands, with 29 bands at 1000 m spatial resolution, five bands at 500 m and two bands (Red and NIR) at 250 m (Esaias et al., 1998). Terra’s orbit around the Earth is timed so that it passes from north to south across the equator in the morning, while Aqua passes south to north over the equator in the afternoon (Guenther et al., 2002; Justice and Townshend, 2002). NASA Ocean Colour Team process the captured data by using a set algorithms to derive land and ocean products including estimates of light attenuation in the water (attenuation of 490 nm wavelength light—\( k_d \)) and phytoplankton abundance (chlorophyll a) (Ouzounov et al., 2004). The ocean colour algorithms are based on radiative transfer theory for light passing through the atmosphere, water and air interface for case 1 and case 2 water columns (Carder et al., 2003). The data is available free on the internet (http://oceancolor.gsfc.nasa.gov) at different spatial and temporal resolutions (Carder et al., 2003; Ouzounov et al., 2004). The current study used light attenuation (\( k_d \)) and chlorophyll a products collected with a spatial resolution of 4 km and averaged over the week (13–20th March) which coincided with the field sampling. The weekly product was the most appropriate for the current study as weather conditions (no rain, strong winds and partial sun) were relatively constant throughout the study period. Daily products were not suitable due to the presence of cloud cover (which influences the image quality) at location coinciding with that days field sampling sites.

2.3. Water quality parameters

2.3.1. Descriptive water physical parameters

2.3.1.1. Temperature and salinity. Temperature (°C) and Salinity (ppt; converted from conductivity (mS cm\(^{-1}\))) were measured to characterise the water bodies present during the current study and identify the importance of freshwater inputs in the near shore coastal waters. These parameters were measured at fixed stations using YSI 6600 Multi-parameter Probe in conjunction with a YSI MDS 650 Handheld Unit. Temperature and salinity were also measured continuously along all transects from the surface to bottom waters using the Biofish, which was tested with factory calibration standards for temperature and conductivity prior to field deployment.

2.3.1.2. Nutrients. Total nutrients (TN and TP) were analysed from unfiltered water samples, and dissolved nutrient samples (FRP and DIN) were collected by filtering water samples through a 0.45 μm membrane filter. The 60 ml syringe and sample bottles were rinsed three times prior to sample collection with site water. All samples were stored in a freezer (<4°C) prior to analysis by Queensland Health Scientific Services, a National Association of Testing Authorities (NATA) accredited laboratory.

Filtered samples were analysed for FRP and NO\(_3\) simultaneously using an automated LACHAT 8000QC flow injection analyser (FIA) using photochemical methods. Unfiltered samples were analysed for total nitrogen (TN) and total phosphorus (TP) using a simultaneous persulfate procedure at 121°C with an initial pH of 13 and a final pH of about 2. If this digestion method did not fully digest all sediment bound nutrients, a Kjeldahl procedure was used to digest the sample. After digestion, analyses for TN and TP were performed using the FIA and photochemical methods (APHA, 1998).

2.3.2. Chlorophyll a

Chlorophyll a (chl a) was used as an estimate of phytoplankton biomass in the water column and was measured in three ways: (1) as extractable chl a collected on glass fibre filters from surface water; (2) using a fluorescence detector on both the YSI probe (surface and depth profiles at fixed stations) and the Biofish (depth profiles of chl a along all transects); (3) using chlorophyll algorithms applied to the MODIS imagery.

2.3.2.1. Extractable Chl a. A known quantity of water was filtered through a Whatman 0.7 μm glass microfibre filter paper under suction. The amount of water filtered was dependent on the turbidity of the water and ranged from 0.7–2.0 L. The filter paper was placed into a graduated screw cap polypropylene tube with 0.01 g of magnesium carbonate which acts as a buffer during the extraction process. The tubes were immediately wrapped
in aluminium foil to block all light and placed in a freezer (−4 °C). The chl a extraction process was carried out in accordance with the Standard Methods for the Examination of Water and Wastewater (APHA, 1998), at the EPA laboratories following standard QA/QC procedures (EHMP, 2004; http://www.healthywaterways.org).

2.3.2.2. Fluorescence probes. At fixed stations chl a was measured as fluorescence and converted to a concentration of chl a (µg L⁻¹) using the YSI 6600 Multi-parameter Probe in conjunction with a YSI MDS 650 Handheld Unit. At the beginning and end of each transect, the probe was lowered into the water and readings were taken at the surface (approximately 50 cm deep), mid water column and just above the bottom. At midtransect stations, a bucket of water was collected from the side of the vessel and the probe inserted into the bucket for readings. The probe was calibrated using laboratory standards at Queensland Environmental Protection Agency (QLD EPA) prior to the field trip (http://www.healthywaterways.org).

Continuous underway sampling of chl a, with the Biofish, also used a fluorescence probe (Seapoint Sensors Inc., USA) and converted to µg L⁻¹. The chl a to fluorescence conversion had been determined using laboratory extracted chlorophyll samples collected from various water depths (previous field trip) as well as surface water samples collected during a night transect to ensure that surface quenching of fluorescence was absent (current field trip). During the day, fluorescence quenching of surface waters was evident in most transects and dependant on light intensity. An estimate of unquenched chl a was determined using an exponential light correction >50 µmol quanta m⁻² s⁻¹, such that surface values were similar to values at the depth corresponding to 50 µmol quanta m⁻² s⁻¹ using data from well mixed waters. This enabled the chl a dataset to be averaged with and without surface quenching for comparisons to other datasets.

2.3.3. Remote sensing. Standard, readily available MODIS chl a maps were used for this study. The chl a maps are generated by the NASA Ocean Colour Team from MODIS imagery and downloaded from the internet. The algorithm to calculate chl a is based primarily on methods and algorithms developed for the CZCS (Coastal Zone Color Scanner) program described by Gordon and Clark (1980) and refined and adapted to the MODIS bands (Carder et al., 2003). The refinement resulted in an improved chl a derivation for coastal waters (case 2) (Carder et al., 2003). For the fixed stations, approximate chl a values were derived from the MODIS chl a map using remote sensing image processing software ENVI 4.0. If the fixed station was located on an image pixel defined as land, the adjacent pixel value that was not influenced by land, was used.

2.3.3. Water clarity

Water clarity was measured using a total of seven techniques that can be divided into three major categories: (1) filtering suspended solids; (2) using probes to estimate concentrations of particles in the water (turbidity probe on the YSI and a beam attenuation probe on the Biofish); (3) estimating light attenuation using 4 different techniques; Secchi disk depth, down welling irradiance profiles using a spectrometer, change in photosynthetically active radiation (PAR) with depth measured with the Biofish and remote sensing.

2.3.3.1. Total suspended solids. A known quantity of water was filtered through a pre-washed and weighed Whatman 0.7 µm glass microfibre filter paper under suction. The amount of water filtered was dependent on the turbidity of the water but ranged from 0.6–4.0 L. Filters were rinsed with distilled water, whilst still under suction, to remove salts and then placed in small vials in the freezer until analysed. Filters were analysed by drying at 65 °C oven and then weighing to determine total suspended solids concentration in mg L⁻¹. The filters were prepared and analysed by the QLD EPA following standard QA/QC procedures (EHMP, 2004; http://www.healthywaterways.org).

2.3.3.2. Turbidity YSI 6600. A YSI 6600 Multi-parameter Probe and MDS 650 Handheld Unit were used to measure turbidity in Nephelometer Turbidity Units (NTUs). The YSI probe was lowered into the water at the beginning and end of transects and readings were taken at approximately 50 cm depth, mid water column and just above the substrate. At sites in the middle of transects, when the vessel was underway, the probe was inserted into a bucket of site water collected from the surface (before any particles had the opportunity to settle). The probe was calibrated using laboratory standards at QLD EPA prior to the field trip (EHMP, 2004; http://www.healthywaterways.org), but not during sampling.

2.3.3.3. Beam attenuation. The beam attenuation coefficient of particles (cₚ, m⁻¹) was calculated from the transmittance (T) of 660 nm light through a 25 cm pathlength (PL) using the equation of Bartz et al. (1978); cₚ = Ln(T)/PL. This is different to a standard turbidity probe in that a turbidity probe measures the amount of light at 660 nm that is scattered, while the beam attenuation probe measures the amount of light that passes through 25 cm of water in a straight line. Although the transmissometer (ADM, Germany) was calibrated to pure distilled water prior to field sampling, values in offshore oceanic waters exceeded 100%. So all data was re-calibrated on the assumption that these deep offshore water samples were optically clear (100% transmittance) and
therefore the true absolute values may be slightly higher
than the reported values.

2.3.3.4. Light attenuation based on Secchi depth. Secchi
depth (SD) (Preisendorfer, 1986) was used as an indica-
tion of light attenuation into the water column and was
only measured at the beginning and end of transects,
when the vessel was stationary. The Secchi disc, a
300 mm weighted, black and white disc was lowered into
the water column and the depth at which the disc was no
longer visible was recorded as the SD. The SD was con-
verted to a SDS value using the following equation
Kd = 1.7/SD as this relationship has been found to be
the most robust over a wide range of turbidities (Carru-
thers et al., 2001).

2.3.3.5. Light attenuation based on downwelling irrad-
iance profiles. A spectrometer (Trios RAMSES) is a
hyperspectral sensor that was used to measure the
downwelling irradiance profiles (W m⁻²) between 320
and 950 nm wavelengths (UV/VIS range), at 3.3 nm
spectral intervals and 0.3 nm bandwidth. The irradiance
profiles were used to determine the attenuation coeffi-
cient at 490 nm (similar wavelength used by MODIS
algorithm to estimate attenuation) as well as being
pooled between 450 nm and 700 nm to give an average
attenuation in the PAR range. At each fixed station
three replicate irradiance profiles were obtained above
the surface, subsurface, 5 m and 10 m depth. The down-
wellling spectral irradiances for each depth were aver-
age to account for any variation in surface light
availability that occurred during sampling. Light atten-
uation was calculated from the linear regression of the
natural log of the average downwelling irradiance spec-
tra at the subsurface, 5 m and 10 m depths (Carru-
thers et al., 2001).

2.3.3.6. Light attenuation based on Biofish. Continuous
underwater light data was measured along each transect
using a 4 pi quantum sensor (LiCor, Nebraska). A 4 pi
quantum was used to minimize the effect of instrument
pitch on descent and ascent. Light attenuation was
calculated from the linear regression of the natural
log transformed PAR profile for light data above
10 µmol quanta m⁻² s⁻¹ (r² > 0.95). A surface light sensor
4 pi (with cosine correction plate—LiCor, Nebraska) was
used to monitor ambient fluctuations during profiles.

2.3.3.7. Light attenuation based on remote sensing. Ocean
water light attenuation properties are estimated by
MODIS using algorithms based on absorbance at
490 nm (Carder et al., 2003). For the fixed stations,
approximate light attenuation values were derived from
the MODIS light attenuation map using remote sensing
image processing software ENVI 4.0. The method used
to exclude land pixels coincident with fixed stations for
chl a was also applied to ensure water pixels were com-
pared to fixed sample station measurements of light
attenuation.

2.4. Sampling technique comparisons

The three sampling techniques used in the current
study (fixed station, underway sampling and remote
sensing) measured different characteristics of the water
to estimate similar parameters that relate to phytoplank-
ton abundance (chl a) and water clarity (TSS, Turbidity,
Kd). However, the spatial scale at which each technique
measures is very different, hence, all comparisons be-
tween different techniques were conducted at the fixed
stations that were sampled during the day, as these rep-
resented the smallest common spatial scale (between 5
and 33 sites). Data from the underway sampling was
an average of the water quality recorded from the top
10 m of the vertical profile at the beginning and end of
each transect as well as at the mid-transect stations. Re-
ome sensing data was estimated using the values associ-
ated with the pixel (4 km × 4 km) closest to the fixed
station or the adjacent pixel, when the fixed station
was located too close to land. Not all techniques were
able to collect data from each fixed station (n = 35) for
various reasons (MODIS could not sample near shore
sites, RAMSES and Secchi could not always be de-
ployed due to rough seas and low sun angle), hence
the number of sites used for method comparison ranged
between five and thirty three locations.

3. Results

3.1. Descriptive water physical parameters

3.1.1. Temperature and salinity

Temperature and salinity were measured using the
YSI probe at fixed stations and the Biofish continuously
along the 200 km of transects between the surface and
the bottom waters. Readings made using the YSI and
Biofish were compared at sites where both were taken
and correlated well (r² = 0.85 temperature; r² = 0.87
for salinity (n = 53)), hence, the more extensive data
collected using the Biofish is presented.

Over the 125 km of coastal waters from Hayman Is-
land south to Mackay Harbour the temperature was rel-
atively constant and high, with an average temperature
of 28.806 °C and less than 1 degree (0.727 °C) variation
(Fig. 2). Salinity was also high and relatively constant,
with an average of 35.456 ppt and range of less than 2
salinity units (1.390 ppt) (Fig. 2). This identified that the
water body, during the sampling period, was a well-mixed system with minimal density gradients in ver-
tical structure. There was a slight increase in salinity ob-
served in the near shore waters, relative to the more
oceanic water at the eastern end of each transect and in the more southerly transects, compared with the northern transects. These results suggest that evaporation rates and the residence time of coastal waters are the dominant factors influencing salinity during the sampling period. The only possible evidence of freshwater input occurred at the end of transect 5 close to the mainland (salinity reduced from 36.3 to 36.0 in the last 5 km).

### 3.1.2. Nutrients

Concentrations of dissolved inorganic nutrients were very low across all sites. Dissolved nitrogen oxides (NO$_x$) were below the limit of detection (0.002 mg L$^{-1}$) at all sites and filterable reactive phosphorus (FRP) was below the limit of detection (0.002 mg L$^{-1}$) at all sites except one. Total N and P were present in higher concentrations, ranging from 0.008–0.021 mg L$^{-1}$ P and 0.10–0.18 mg L$^{-1}$ N (Fig. 3). With the highest concentration of both TN and TP observed closest to the mainland.

A positive correlation existed between extractable water column chl $a$ and total N ($r^2 = 0.77$) and total P ($r^2 = 0.75$) (Fig. 3). This trend was particularly evident at sites with chl $a$ concentration above 1 μg L$^{-1}$ chl $a$. There was also a significant correlation ($r^2 = 0.79$) between total P and turbidity (NTU), with the sites closest to the mainland having the highest values of both.

### 3.2. Chlorophyll $a$

Chlorophyll $a$ concentrations measured as extractable chl $a$, after filtering the water, ranged between 0.5 and 2.5 μg chl $a$ L$^{-1}$ (Fig. 4), with the lower concentrations occurring distal to the coast (near oceanic water) and the higher concentrations occurring closer to the mainland. The concentration of chl $a$ measured using the other techniques (in-situ fluorescence and remote sensing) followed a similar trend with the highest concentrations being recorded near the coast with all techniques (Fig. 5). The range in the absolute chl $a$ concentration was largest when measured with the YSI (0.1–4 μg chl $a$ L$^{-1}$) and smallest when measured using either the Biofish (0.5–1.5 μg chl $a$ L$^{-1}$) or MODIS (0.3–1.4 μg chl $a$ L$^{-1}$). The only technique that demonstrated occurring distal to the coast (near oceanic water) and the higher concentrations occurring closer to the mainland. 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strated a significant correlation with the extractable chl a concentrations, was the fluorescence probe on the Biofish ($r^2 = 0.82$, Fig. 4).

The Biofish data demonstrated that the raw fluorescence measurements of chl a near the surface were often reduced relative to those deeper in the water column (Fig. 6a). In clear oceanic water there appeared to be quenching of the fluorescence signal down to 20 m, with this rapidly reducing in turbid water. Quenching of the chl a fluorescence signal appeared to begin at light levels of 50 $\mu$E (µM) and intensified exponentially above this quantity of light. On average, at light levels of 500 $\mu$E, fluorescence values were three times less then they should have been, assuming a vertically mixed water column. This was assumed to be due to quenching of chl a fluorescence. Hence, the chl a data used for comparison purposes in the current study have been corrected using the following equation: $y = 0.923e^{0.0023x}$, (where $x$ is the light intensity at a specific depth, $r^2 = 0.99$; Figs. 5–7). The assumption that the observed depression in chl a

![Fig. 6. Biofish transects: (a) Uncorrected chlorophyll a (µg L$^{-1}$) for day transect 1. (b) Corrected chlorophyll a (µg L$^{-1}$) for day transect 1. (c) Uncorrected chlorophyll a (µg L$^{-1}$) for night transect 1. (d) Uncorrected chlorophyll a (µg L$^{-1}$) for transect 5. (e) Corrected chlorophyll a (µg L$^{-1}$) for transect 5. (f) Beam attenuation of particles (c$_{p,m}$ m$^{-1}$) for transect 5. (g) Light (PAR) at various depths (µEm$^{-2}$s$^{-1}$) on transect 5.](image-url)
fluorescence in the surface waters was due to light intensity was supported by the fact that it was not present in the transect conducted 2 h after sunset, even though other water quality parameters were similar (Fig. 6c). A similar correction was not attempted with the YSI probe as this did not have a light sensor on it and the quantity of data points was much less making it difficult to determine an appropriate correction.

The large pixel sizes of the MODIS image made it difficult to get accurate estimates of \( \text{chl} \) near shorelines (Fig. 8a). For nine stations the underlying pixel value was used. While for another nine stations the \( \text{chl} \) value of an adjacent pixel was used as it appeared to represent a water body with similar characteristics. This allowed semi direct comparison between in-situ measurements and MODIS data at 18 of the 19 fixed stations. The MODIS data was not as closely correlated with the extractable \( \text{chl} \) measurements as the Biofish data, yet it still demonstrated a similar trend as the extractable \( \text{chl} \) data and was rarely more than a factor of two different from the extractable \( \text{chl} \) concentrations (Fig. 4), a relatively close result considering the large difference in spatial resolution of the two data sets.

### 3.3. Water clarity

#### 3.3.1. Turbidity

The total suspended solid concentrations (TSS) measured at the fixed stations (\( n = 33 \)) ranged from 2.2 mg L\(^{-1} \) to 24 mg L\(^{-1} \). However, only 12% of the sites...
and muddy sediment. There were strong correlation between all the techniques used for measuring turbidity (TSS, Beam attenuation, NTU; Fig. 9 \( r^2 = 0.95 \) and 0.88, respectively). However, the data set was skewed towards lower values. Hence, the four sites with values >10 mg L\(^{-1}\) had a greater influence on the correlation than the lower values. When the correlation was calculated removing the values >10 mg L\(^{-1}\), the linear regression was similar, but the amount of variation explained by the correlation reduced.

All three measurements of turbidity (TSS, Beam attenuation, NTU) when correlated against light attenuation \((k_d)\) measured with the RAMSES, between 450 nm and 700 nm, showed very high correlations \(r^2 = 0.89\) to 0.98, as did the \(k_d\) values measured using the Biofish \(r^2 = 0.99\); Fig. 10). However, the estimates of \(k_d\) made using the Secchi disc and MODIS data did not correlate with the RAMSES \(k_d\) as closely as the other techniques \(r^2 = 0.81\) and 0.38, respectively), but did show a similar trend (Fig. 10). The MODIS data probably had a weaker correlation than the other data sets as its spatial resolution was much larger than any of the other sampling techniques (MODIS Pixel size 4 x 4 km; Fig. 8b). This meant we were not able to estimate a MODIS \(k_d\) value for the fixed station with the highest \(k_d\) value, due to its proximity to the coastline, reducing the range of \(k_d\) values that the correlation was based on.

### 4. Discussion

Although the current study took place during the end of the “wet season”, there had only been light showers prior to the sampling event occurring and most of the coastal rivers were not flowing. This explains the absence of any significant freshwater signal in the salinity close to the mainland and the likelihood that the observed salinities are due to adjacent Coral Sea water which has a salinity of approximately 35 ppm evaporating in the near shore regions during the hot (summer) weather. This would result in the observed high salinities where the residence time of the water was greatest. This low influence of freshwater is typical for southern lagoon waters (Picard et al., 1977) and suggests that the other water quality conditions observed during the current study are representative of conditions that occur for much of the year in this region, but will not be representative of water quality conditions that occur during major run-off events.

The strong correlation between chl \(a\) and total nutrients (TN, TP) in the water column suggests that even at the near-shore sites, that had high turbidity, phytoplankton are not light limited and supports the use of chl \(a\) and other pigments as a proxy for nutrient availability (Devlin et al., 1999; Paerl et al., 2003). Assuming a chl \(a\) : Carbon ratio of 1:50 (Cloern, 1995) and redfield ratio (C:N:P, 106:16:1) we have estimated that at
2 µg L\(^{-1}\) chl \(a\) (near top of recorded range), phytoplankton only account for approximately 12% of the TN and 8% of the TP. While these numbers would change significantly if either the chl \(a\):Carbon ratio or the C:N:P ratio changed relative to the assumed values, it gives a good indication that phytoplankton biomass is not the dominant form of nutrient in these near shore case 2 waters, as would be expected in more oceanic parts of the GBR which have case 1 waters (Morel, 1977). During major flood events the chl \(a\) in the water column does not correlate with TN and TP (Brodie and Furnas, 1996), suggesting that phytoplankton are probably light limited where total nutrients are high or that they have not had time to respond to the increased nutrient availability.

By sampling chl \(a\) at various depths in a well mixed water column, the current study has shown that significant quenching of the chl \(a\) fluorescence signal occurs during the day in the surface waters, which agrees with observations made by Falkowski and Kolber (1995). Based on the assumption that the water column was well mixed we were able to correct for this quenching by applying a natural log based algorithm based on the observed chl \(a\) and light present at that depth. However, even with this correction algorithm applied, the surface waters often had lower fluorescence values than deeper in the water column, suggesting that phytoplankton are either, not evenly distributed in the water column or that quenching of chl \(a\) fluorescence was greater in the surface waters than the correction allowed for. Irrespective of what mechanism is causing the observed reduction in chl \(a\) fluorescence at the surface during the day, it is clear that any estimate of chl \(a\) based on fluorescence measurements of surface water during the day, has the potential to underestimate the total chl \(a\) concentration present in the water column, unless corrections are made for variation in fluorescence caused by irradiance (Morrison, 2003). The results from the current study suggest that a comprehensive monitoring program based around chl \(a\) abundance should use several different methods of analysis for validation of results and sample from multiple depths, especially if the measurement are based on “in situ” fluorescence measurements.

As there was only a small amount of freshwater input prior to and during our sampling period, it is reasonable to assume that the majority of turbidity observed during the current study was due to wind resuspension (Wolski and Spagnol, 2000) and not from catchment run-off. This is supported by our observations, where the exposure of an embayment to the prevailing S.E winds, water depth and sediment type all appeared to be important in determining the turbidity of the water, this was supported by the highest turbidity observed in the current study, occurring in a bay that was exposed to the S.E winds, approximately 5 m deep and had muddy sediment, with no significant freshwater input at the head of the bay. We also observed that adjacent to islands the gradient of clearer water in the east and dirty water in the west was often reversed, suggesting that resuspension of island sediments is an important factor in determining both the turbidity and nutrient availability in the water column both adjacent to and distant from the mainland. This aspect of GBR water quality, where it is dominated for the majority of the year by prevailing weather conditions and influenced by catchment run-off for relatively short periods could be wrongly interpreted to suggest that catchment run-off is not important in determining long term water quality. However, it is actually the deposition and resuspension of sediments and their associated nutrients, delivered to the coast during run-off events, that continue to influence the water quality of near-shore waters during dry periods (Brodie and Furnas, 1996; Devlin et al., 2001a).

The close correlation observed in the current study between the different methods of measuring water clarity (TSS, turbidity, and \(k_d\)), suggest that conversion algorithms could be developed for different methods. This would enable water clarity measured using different methods to be converted to a common unit for comparison throughout the GBR. This means that data collected by community groups using relatively low tech measurements (e.g. TSS, NTU, Secchi) can be used in conjunction with more technical water quality sensors and remote sensing to provide a complete spatial and temporal coverage of the GBR. The two estimates of water clarity that had the weakest correlations with the other techniques were Secchi and MODIS estimates of \(k_d\). The fact that some Secchi readings were shallower than predicted by the \(k_d\) calculated using the light meter was probably due to the poor sampling conditions (i.e. partial cloud cover, 2+ m seas and strong tidal currents; Preisendorfer, 1986), while the poor correlation of the MODIS data is primarily due to the different spatial scale (1 m\(^2\) for fixed station and 16 km\(^2\) for MODIS) at which MODIS estimates water quality parameters and the fact that MODIS could not be used adjacent to the land where the largest light attenuations occurred. This had the effect of reducing the range of \(k_d\) values over which the correlation was calculated. However, the \(k_d\) values obtained from the MODIS were in a similar range as those determined by the other techniques (e.g. RAMSES) and similar to light attenuations observed for other near-shore systems with a similar gradient in water quality (e.g. Moreton Bay; Phinn et al., this issue).

The current study used remote sensing data that was freely available and easily downloaded from the web (4 km pixel from MODIS; Carder et al., 2003). However, other satellite products are being developed with increased resolution, for example specific algorithms
Table 2
Comparison of the spatial and temporal scale at which various sampling techniques can be applied and their relative advantages and disadvantages

<table>
<thead>
<tr>
<th></th>
<th>Remote sensing</th>
<th>Underway monitoring</th>
<th>Fixed stations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Description</td>
<td>• Image of water surface taken from satellite&lt;br&gt; • Two dimensional (horizontal)</td>
<td>• YSI probe (underway)&lt;br&gt; • One dimensional</td>
<td>• Buoy with WQ sensor deployed at fixed stations&lt;br&gt; • One dimensional&lt;br&gt;</td>
</tr>
<tr>
<td>Indicators</td>
<td>• Temperature, light attenuation ($k_d$ at 490 nm), chl $a$</td>
<td>• Temperature, salinity, NTU, DO, chl $a$</td>
<td>• Temp, salinity, turbidity&lt;br&gt; Nutrients, toxicants, Secchi depth, species composition and spectral absorbance/reflectance plus water quality parameters</td>
</tr>
<tr>
<td>Example</td>
<td>• MODIS example from this study</td>
<td>• Hodge et al. (this issue)</td>
<td>• EPA Autonomous samplers&lt;br&gt; EHMP (EHMP, 2004; <a href="http://www.healthywaterways.org">www.healthywaterways.org</a>)&lt;br&gt; Current study</td>
</tr>
<tr>
<td>Application</td>
<td>• Continuous coverage of large areas&lt;br&gt; • Regular monitoring possible&lt;br&gt; • Rapid, simple, cheap assessment&lt;br&gt; • Results can be compared to many other global studies on a daily basis</td>
<td>• Continuous measurement of surface WQ parameters&lt;br&gt; • Ideal to identify discontinuities or confirming gradients</td>
<td>• Permanent monitoring to identify temporal changes&lt;br&gt; Traditional WQ parameters and new measures of ecosystem health&lt;br&gt; Scale depends on monitoring requirements&lt;br&gt; Ideal for community participation</td>
</tr>
<tr>
<td>Spatial scale</td>
<td>• Global cover&lt;br&gt; • 16 km$^2$ or 1 km$^2$ resolutions available free&lt;br&gt; • Better resolutions available at cost</td>
<td>• Continuous measure along surface transect&lt;br&gt; • Can cover large areas (&gt;100 km$^2$)</td>
<td>• Restricted to specific sites&lt;br&gt; Only samples a small amount of the area of interest</td>
</tr>
<tr>
<td>Temporal scale</td>
<td>• Daily&lt;br&gt; • Dependant on cloud cover and resolution required</td>
<td>• Variable&lt;br&gt; • Dependant on boat and crew being available</td>
<td>• Excellent temporal sampling&lt;br&gt; Accurate measurement at one point in time</td>
</tr>
<tr>
<td>Sampling and processing time</td>
<td>• Daily snap shot possible&lt;br&gt; • Setup required to handle large data sets and undertake interpretation of the images</td>
<td>• Data processed instantaneously&lt;br&gt; • Additional time required to QA data, calibrate and service sensors and for interpretation</td>
<td>• Instantaneous&lt;br&gt; Additional time required to QA data, calibrate sensors and for interpretation&lt;br&gt; Sampling quick with extensive processing time following&lt;br&gt; Community participation possible</td>
</tr>
</tbody>
</table>
for Queensland waters using the MODIS image with a 1 km pixel size Dekker (pers. comm.). These will improve both the resolution and interpretation of the unprocessed image for Queensland’s waters.

This paper has focused on comparing both qualitative and quantitative data obtained by different sampling techniques. The relative spatial and temporal scales at which these techniques can be applied is summarised in Table 2, with a summary of their advantages and disadvantages. No water quality sampling technique used in isolation can adequately address the monitoring requirements of a region as big as the GBR. However, we hope that the current study has demonstrated how different sampling techniques can be used in conjunction to enhance our understanding of the complex three dimensional aspect of water quality in the GBR lagoon.

The aim for management is to combine sampling techniques, that operate at different spatial and temporal scales, into a monitoring program that is sensitive enough to detect the levels of change likely to occur from management actions in the catchment and is also cost effective to run.

While the optimal combination of techniques will vary depending on the management objectives and priorities, the following guidelines should assist in the design of a new monitoring program for the GBR:

1. Monitoring should involve a multi-scaled sampling approach (as our paper demonstrates), with a focus on key parameters that are expected to respond to management actions. In addition to the techniques presented in the current study it is important to identify relevant, but simple measurements to enhance community participation (Devlin et al., 2001b), and strategic fixed stations to improve our understanding of temporal variability in key parameters such as turbidity and chl a (e.g. Falkowski and Kolber, 1995).

2. It is important to consider the ultimate long term vision or goal and ensure that the techniques used are sensitive enough to detect various milestones in achieving the environmental vision, this will require a strong link between monitoring and modeling to provide the ability to predict perturbations in water quality that are likely to occur in the future and have occurred in the past in conjunction with an error or variability prediction.

3. Integration with existing and planned catchment and GBR monitoring programs to reduce the time required before trend analysis can be undertaken and enhance our understanding of cause and effect (e.g. event based monitoring).

4. Improved integration between different sampling techniques, so that each technique contributes to reducing the error in the assessment of water quality.
Acknowledgments

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References


