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Research papers

Application of a water balance model using depth measurements in the Mungalla wetland in north Queensland, Australia

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ABSTRACT

The flows of water into and out of wetlands are primary determinants of their ecological condition, habitat suitability and potential for removing pollutants. However, comprehensive measurements of the complete water balance are rarely made due to monitoring complexities and associated costs. The water balance model described here has the advantage that it can provide an estimate of the main inflows and outflows to and from a wetland without the need for their direct measurement. The only daily variables required are weather data and the wetland depth, which is easy and inexpensive to measure with simple loggers. The model is applied in the Mungalla wetland, north Queensland, Australia using a unique 10 years of daily depth measurements. Modelled inflows and outflows are shown to be highly variable within and between years, according to the rainfall inputs. This illustrates the highly dynamic nature of these wetlands which has implications for their ecological condition. The model has also been applied in two other wetlands in Queensland (Wallace and Waltham, 2021; Wallace et al., 2022), demonstrating its potential application in many more wetlands. We also discuss the potential use of the model to determine aquatic risk periods and the estimation the nitrogen and sediment removal of different wetlands. This approach will be very valuable in helping evaluate the type and location of treatment wetlands on coastal floodplains that can make the best contribution to load reductions to the Great Barrier Reef lagoon.

1. Introduction

Coastal zone wetlands in north Queensland, Australia play an important role in providing ecosystem services such as biodiversity, water quality improvement as well as cultural and community services (Adame et al., 2019a; Canning et al., 2023; Arthington et al., 2015). However, land-use changes within catchments and major modifications to floodplains (e.g. urbanisation and agricultural expansion), like so many other places around the world, have contributed to the degradation and loss of wetland habitats (Waltham and Sheaves, 2015; Waterhouse et al., 2016; Davis et al., 2016). Since the European settlement (~1850) in Queensland, land-use changes within catchments and major modifications to floodplains have contributed to the degradation and loss of wetland habitats. In the Great Barrier Reef (GBR) catchments, between 78 % and 97 % of wetlands that existed prior to settlement remain, however, this varies among regions and catchments, and the losses are more substantial for some wetland types in certain locations (e.g. more palustrine compared to estuarine wetlands have been drained and replaced with agricultural development) (Furnas, 2003; Great Barrier Reef Marine Park Authority, 2014; Sheaves et al., 2014; Waltham and Sheaves, 2015; Canning and Waltham, 2021). Many of the wetlands that remain are degraded for a number of reasons. These include; earth bunding to exclude seawater and reclaim land for pasture (Great Barrier Reef Marine Park Authority, 2014; Abbott et al., 2020); upstream agricultural use (grazing and sugar cane production) which can leach ecologically damaging nutrients and sediments (Arthington et al., 2015; Pearson et al., 2013; Adame et al., 2019b; Waltham et al., 2021), and extensive aquatic invasive weed chokes (Butler et al., 2009; Burrows and Butler, 2012; Waltham and Fixler, 2017), which can lead to hypoxic conditions and fish kills (Flint et al., 2015; Perna et al., 2012). All of these impacts can render coastal floodplains severely compromised in terms of the ecosystem services they can provide.

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In response to global biodiversity and environmental conservation and protection targets, there is increasing interest in halting this degradation on the GBR floodplains, and a desire to commence large-scale programs to repair and restore wetland ecosystem habitats (Adame et al., 2019a,b; Waltham et al., 2019). An additional benefit of this is seen as the potential for wetlands to remove nutrients and sediment from upstream anthropogenic sources that can negatively impact aquatic ecosystems on both the coastal floodplains and within the GBR lagoon (Adame et al., 2021; Kavehei et al., 2021; Wallace and Waltham 2021; Wallace et al., 2022). However, wetlands are dynamic ecosystems that vary over space and time. Their water content is driven by seasonal rainfall and activities on the land and their ability to improve water quality is highly variable. There are currently very few comprehensive studies of wetland hydrology in the GBR catchments where surface and groundwater flows are directly estimated (Waltham, 2023a). A rare example is the three year study of a natural riverine wetland in the Tully catchment in north Queensland (McJannet et al., 2012a,b). Here surface flow measurements and groundwater flux estimates were combined with water nutrient and sediment concentration measurements to find that this wetland removed little or no nitrogen and sediment. Another study of a constructed wetland near Babinda, north Queensland showed that it removed 37 % of the nitrogen and 62 % of sediment entering that wetland during flood pulses (Wallace and Waltham, 2021). A further constructed wetland near Baker Creek, Mackay removed 52 % of its nitrogen and 86 % of its sediment load (Wallace et al., 2022). Clearly, the efficacy of wetlands in removing nitrogen is very variable and highly dependent on the wetland type and location, which determines its hydrological characteristics. More comprehensive studies where surface and groundwater flows and water quality are measured in a range of wetlands in different locations are therefore needed in order to determine the best wetland types and locations for maximum water quality improvement. However, the number of wetlands that can be monitored in this way will be limited by funding and availability of appropriate skills, so there is a need to derive a method to quantify the water balance of wetlands in a simpler and less expensive way (Twomey et al., 2024). In addition, wetlands can have multiple and/or diffuse inlets and outlets which cannot easily be monitored directly. To help address this issue this paper describes a method where a wetland water balance is derived from measurements of water depth and readily available weather data. The model developed here uses one of Australia's longest wetland data sets, a long term (10 years) depth measurement campaign made in the Mungalla wetland, north Queensland. The model estimates daily values of water inputs to the wetland (as run in and direct rainfall input) and water losses (as drainage and evaporation). These are combined to derive the seasonal dynamics and inter-annual variability of this wetlands' water balance. The water balance model also has important applications in that it can be used to identify periods of aquatic risk in the wetland, along with the possibility of combining it with a nutrient balance model which then allows nitrogen and sediment removal to be quantified. The modelling approach presented here will help managers optimise strategies to increase conservation and restoration of coastal wetlands, which represents a major step forward in delivering on national biodiversity and environmental protection targets.

2. Location and climate

The semi-permanent Mungalla wetland complex covers an area of ~ 160 ha within the Mungalla Station (18°42′21″S, 146°15′34″E), an 830 ha property located in the lower part of the Herbert River catchment south east of Ingham, north Queensland (Fig. 1). Mungalla station was acquired by the Nywaigi Aboriginal Land Corporation in 1999 and has been operating since 2001 by the Mungalla Aboriginal Corporation for Business, who (among other ventures) run a cattle grazing enterprise, with a combination of agisted and owned stock. The wetlands within this property are bounded to the west by grazing lands and to the east by regrowth forest on coastal sand ridges. There are mangrove and other saltmarsh wetlands along the coast and to the south of the property, which is adjacent to the Great Barrier Reef lagoon. Inland, the surrounding catchment is dominated by sugar cane farms with some areas of grazing. In the early 1900′s an earth bund was constructed to exclude seawater which created a ponded pasture for grazing (Grice et al., 2012; Abbott et al., 2020). Over time, this led to extensive freshwater weed

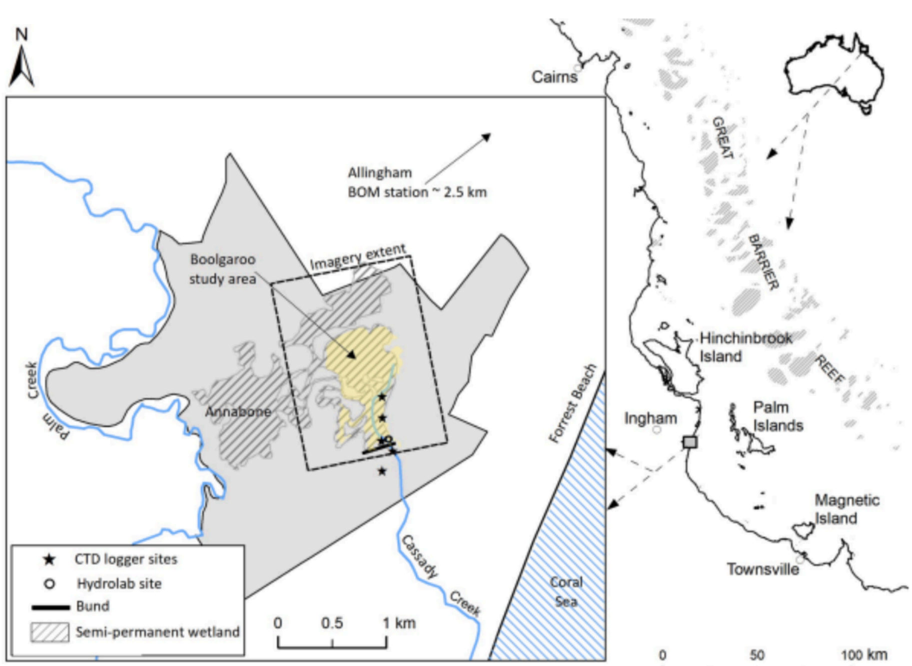

Fig. 1. The location of wetlands within Mungalla station in the lower Herbert River catchment, Queensland, Northern Australia. The Mungalla wetland complex (160 ha) is hatched in dark grey with the Boolgaroo sub-region (60 ha) of the wetland shown in yellow. Also shown are the locations of the logger sites above and below the earth bund which was removed on 6th October 2013.
Reproduced from Abbott et al (2020).

infestations, so the earth bund was removed at the beginning of October 2013. The effect of this on drainage from the wetland is presented in this paper. Other effects on seawater entering the wetland on very high tides and the impact of this on the wetland vegetation are described elsewhere (Abbott, et al., 2020).

The study region has a wet tropical climate with highly variable seasonal and annual rainfall, with the mean (2080 mm; 1968 to 2016) at nearby Ingham and is strongly seasonal with 85 % falling in the six wettest months, November to April. Because of the highly seasonal rainfall freshwater mainly enters the wetland in the wet season as direct rainfall input, runoff from the surrounding sub-catchments and overbank flow from Palm Creek, which runs along the western boundary of the wetland. Ambient air temperatures are highest in December (daily average 27.3 °C) and lowest in July (daily average 19.3 °C), with high humidity (~63–77 %) throughout the year.

Wetland water depth, temperature and electrical conductivity (used to detect salinity) were monitored by loggers (CTD-Diver, Eijkelkamp Soil & Water, The Netherlands) located in five permanent positions in the wetland, beginning on 24 October 2012. The locations are 450 m, 250 m and 50 m above the bund location and also 50 m and 250 m below it (Fig. 1b). The loggers capture data from the bottom of the water column (~ 5–10 cm above the soil surface) every 15 min and these are downloaded every month during routine service visits. Ancillary data used in conjunction with the above wetland data are daily rainfall measured at Allingham (Australian Bureau of Meteorology station No 032117).

3. Water balance model

The change in depth (δd) of the wetland on any day is given by the difference between water entering and leaving it. This can be expressed as;

$$\delta d = (P + Rin) - (Dw + Ew), \tag{1}$$

where P is the rainfall directly entering the wetland, R_{in} is the water which flows into the wetland from its surrounding catchment, D_w and E_w are the drainage and evaporation from the wetland respectively. There were a few occasions when seawater entered the wetland on very high spring tides (Waltham et al., 2023b). However, the water level rise was very small compared to the depth changes caused by rainfall and run-in, so they did not contribute very much to the overall water balance. This paper therefore focusses on the freshwater balance of the wetland, with the frequency and duration of seawater ingress and its ecological impacts presented elsewhere (Abbott et al., 2020; Karim et al 2021)

3.1. Run in

Estimates of the amount of water that flowed into the wetland from its surrounding catchment (R_{in}) during rainfall were made using a simple runoff coefficient model which assumes R_{in} is a fixed fraction of rainfall, C (e.g. see Pilgrim and Cordery, 1993).

$$Rin = C^*P \tag{2}$$

As small amounts of rainfall do not generally produce runoff due to losses from interception and depression storage (Critchley and Siegert, 1991), only events > 5 mm were used to calculate R_{in}. Above this threshold the value of R_{in} is also affected by the wetness of the surrounding catchment, with less runoff occurring when it is dry. To account for this in a simple way we calculated daily values of the soil moisture deficit (SMD_c) in the surrounding catchment as the difference between rainfall (P) and catchment evaporation (E_c). Unless the catchment is saturated, E_c will be less than the wetland evaporation, E_w, and mainly controlled by the soil moisture deficit and following Shuttleworth (1993) is given by,

$$Ec = KcEw \tag{3}$$

where K_c is a 'crop coefficient' which is a function of the soil moisture deficit (SMD_c). The form of this relationship is shown in Fig. 2, where soil moisture does not reduce E_c until $SMD_c = 100$ mm, after which K_c decreases linearly until it reaches zero at the maximum soil moisture deficit SMD_{max} of 200 mm; typical of a 2 m deep sandy loam soil (Burk and Dalgliesh, 2013). Runoff is then recalculated as

$$Rin = Rin\left(1 - \frac{SMDc}{SMDmax}\right) \tag{4}$$

The value of the catchment runoff coefficient, C was estimated by optimization for the years 2013 and 2104. Values of D_w and E_w on each day were calculated as described in the sections below. The root mean square error (RMSE) between the modelled (using equation (1)) and measured daily depths was minimized by iteratively altering the value of C. The effect of this on the model predictions of the rise in the wetland water depth during the first five large ($P > 100$ mm) rainfall events in 2013 and four events in 2014 are shown in Fig. 3.

The regressions shown in Fig. 3 are for when $C = 3.03$, and the correlation between the model and observed depths were high in both 2013 ($r^2 = 0.99$) and 2014 ($r^2 = 0.97$).

The R_{in} estimation method described above only generates run-in on days when it is raining. However, run-in can continue for a number of days after rainfall, especially for large events. To account for this we have distributed the total run-in from any given event over the rainfall day and 10 days following the rainfall. This was done using an exponential decay function of the form:

$$Rin = P(1 - b)^t \tag{5}$$

where the daily rainfall (P) generates run-in (R_{in}) for several days (t) after the rain event. The rate of decline of R_{in} is set by the value of the decay coefficient b and this is shown in Fig. 4. We also obtained the value of $b = 0.5$ by minimizing the root mean square error (RMSE) between the modelled and measured daily depths.

3.2. Evaporation

The wetland evaporation rate was estimated using the energy balance model described by McJannet et al., (2008; 2013)), which was originally developed for calculating daily evaporation from open water bodies of various sizes. The main input of energy to the model is solar radiation and the main losses are via heat conduction to the atmosphere and evaporation. The model requires daily weather data, which were

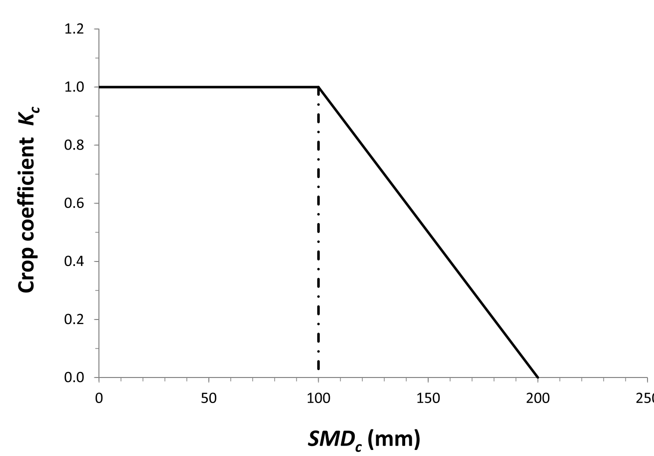

Fig. 2. The relationship between the crop coefficient (K_c) and the catchment soil moisture deficit (SMD_c) (.
adapted from Allen et al., 1998)

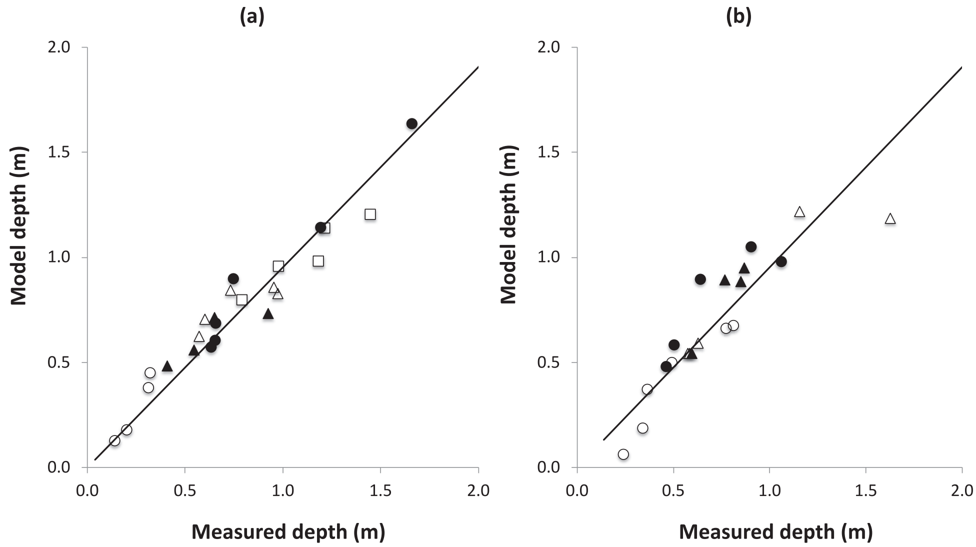

Fig. 3. A comparison of modeled and measured wetland depth for (a) five rising water events in 2013 (y = 0.95x; r^2 = 0.99) and (b) four rising water events in 2014 (y = 0.95x; r^2 = 0.97).

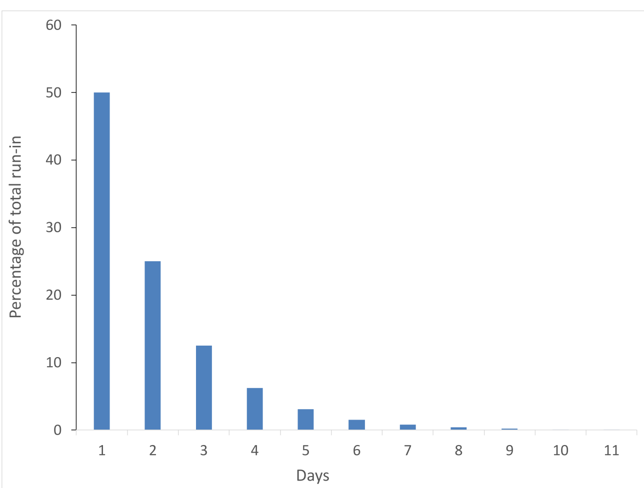

Fig. 4. The distribution of run-in on any rain day (t = 1) and the following days.

obtained for nearby Allingham from the Scientific Information for Land Owners (SILO) database (https://www.nrw.qld.gov.au/silo/). The SILO database consists of interpolated meteorological variables on a 0.05° (5 km) grid for the whole of Australia (Jeffrey et al., 2001). The particular SILO variables used by the wetland evaporation model are air temperature, vapour pressure, solar radiation and rainfall. Wind speed is also required to calculate evaporation, but is not available in SILO, so we used the mean daily wind speed for nearby Ingham, 3.3 m s^{-1} (Australian Bureau of Meteorology Station No 032078). The way these variables are used to calculate daily evaporation are described by McJannet et al., (2008) and Wallace et al., (2015).

3.3. Wetland dry periods

Towards the end of each dry season, water depth in the wetland becomes very low and can eventually reach zero (i.e. the wetland is completely dry). When the water depth is zero the bare soil at the bottom of the wetland becomes exposed and water can evaporate from it leading to the build-up of a moisture deficit in the wetland (SMD_w). Any rain (P) or run in (R_{in}) during the period when the wetland soil is exposed will have to replenish SMD_w before the depth of the wetland can rise above zero. To account for this a soil moisture deficit sub-routine was used to calculate SMD_w in periods when the wetland depth was below 5 cms. This depth was chosen to recognize that with the uneven surface at the bottom of the wetland, some parts of it may become exposed to the air before the measured depth reached zero. Evaporation from the bare wetland soil (E_{ws}) was calculated using the Ritchie (1972) model. This model calculates evaporation in two stages; the first is the potential evaporation rate (taken here as the wetland free water evaporation rate, E_w), which continues up until a threshold amount of water has been lost (U);

$$Ews = Ew \tag{6}$$

Once U has been exceeded, stage 2 begins during which evaporation decreases such that the cumulative evaporation is inversely proportional to the square root of time (t), i.e.

$$\Sigma Ews = \alpha t^{-1/2} \tag{7}$$

where α is a soil dependent constant. Here we set U = 50 mm to allow for the water stored on the uneven wetland soil surface (equivalent to a depth of 5 cms). The value of α was set equal to the values given for a Plainfield sand in Ritchie (1972), i.e., 3.3.

The effect of including soil moisture deficits in the catchment (SMD_c) and wetland (SMD_w) in the model simulation of wetland depth is shown in Fig. 5. During the wet season, when SMD_c is relatively small and there is no SMD_w, modeled depths are similar with and without soil moisture deficits. Towards the end of the dry season, when significant soil moisture deficits develop in the catchment, simulated depths begin to be affected by SMD_c as this reduces run in to the wetland. However, the biggest effect on simulated depth occurs during and after the periods when wetland depth is close to zero. This is because rainfall and any run in during or after this period have to replenish SMD_w before water can start ponding on the surface. This means that the periods when wetland depth is zero can be greatly underestimated if SMD_w and SMD_c are not included in the model. For example, at the end of the 2013 dry season simulated wetland depth is below 5 cms for 125 days with soil moisture deficits included and only 69 days when they are not included. The equivalent figures for the end of the 2014 dry season are 117 and 38 days respectively. This illustrates that the soil moisture deficits in the catchment (up to 200 mm) and within the wetland (up to 278 mm) have a very significant effect on the depth of water within the wetland.

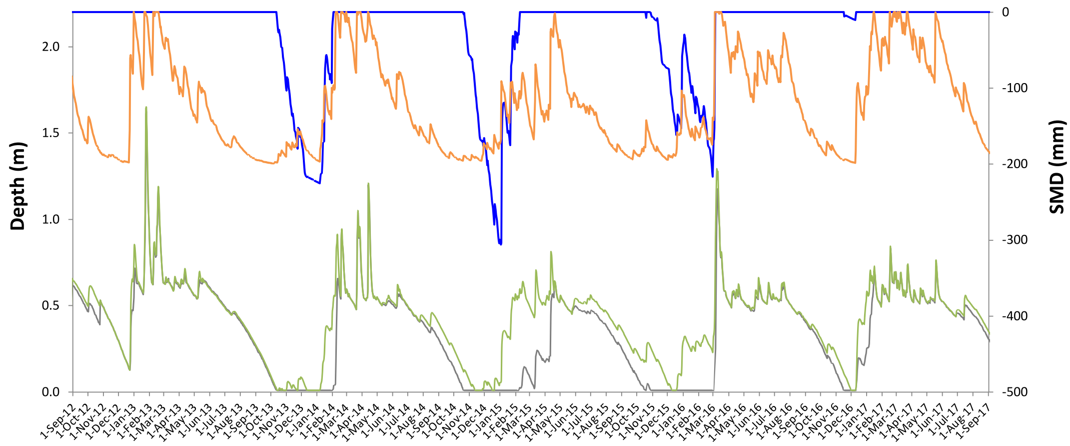

Fig. 5. Seasonal changes in wetland depth modeled with (green) and without (grey) soil moisture deficits (SMD_c and SMD_w). The catchment SMD_c is shown in orange and the wetland SMD_w, when there is no water in it, is shown in blue.

3.4. Wetland drainage

Drainage from the wetland was calculated from the daily decrease in wetland depth recorded in the rain free periods following rain events (see Wallace et al., 2020). As water also evaporates from the wetland during this time, daily drainage estimates are given by the change in water level (δd) minus the evaporation (E_w) on each day. On a small number of the occasions selected ($\sim$ 7 %) we estimated that there was some run-in (R_{in}) to the wetland and where this occurred this was added to the drainage estimate.

Fig. 6 shows the relationship between drainage and wetland depth before and after the bund was removed. When depths were low, drainage was very slow, averaging around 2–3 mm d^{-1} both before and after the bund was removed. As the wetland depth increased, drainage rates increased markedly, with the increase starting at a lower depth after the bund was removed. For any depth $> \sim$ 60 cms drainage was higher after the bund was removed, so separate linear regressions were fitted to the data before and after the bund was present (Fig. 6). The depth above which data were included in the regressions were altered iteratively until the best fit (highest r^2) was obtained; this threshold depth was 65 cms before bund removal and 54 cms after it was removed. The regressions shown in Fig. 6 were used in the wetland water balance model to calculate ‘rapid’ drainage from the wetland when depths were above these two thresholds.

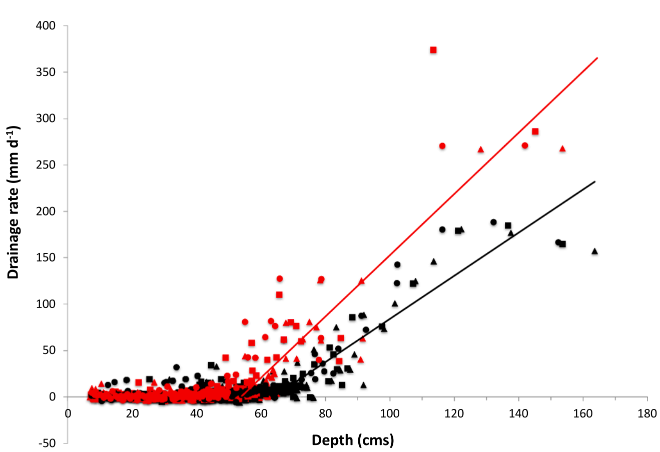

Fig. 6. The wetland drainage rate before (black) and after (red) the bund was removed. Drainage data shown are for the three locations above the bund; 50 m (squares), 250 m (triangles) and 450 m (circles). The straight lines fitted to data have the form; black line, $D_w = 2.32\ d - 148$ ($r^2 = 0.86$) (depth > 65 cms); red line, $D_w = 3.30\ d - 178$ ($r^2 = 0.79$) (depth > 54 cms).

When the wetland depth was less than 54 cms drainage was slow, and individual daily values were highly variable (Fig. 6), due to uncertainties in the daily values of δd and E. Better estimates of the slow drainage rate can be obtained using two distinct drying (and rain and run in free) periods at the end of the dry seasons in 2012 and 2013. In the first of these, from 20 November 2012 to 10 December 2012, the wetland level dropped by 160 ($\pm$ 2.6) mm at the three above bund locations. During the same 21 day period, the total evaporation from the wetland was 148 mm, so the difference ($\delta d - E$), was 12 mm, equivalent to an average drainage rate of 0.6 mm d^{-1}.

Fig. 7(a) shows a plot of modeled versus measured water depth during the period 20 November 2012 to 10 December 2012, assuming a fixed daily drainage rate of 0.6 mm d^{-1}. To eliminate any differences that are not due to drainage, the model depth was set equal to the measured depth at the start of the comparison period. The model reproduced the measured daily depths very accurately ($r^2 = 0.997$), implying a fixed drainage rate that was independent of depth. A more independent test of how the model estimated the slow drainage rate in 2013 is shown in Fig. 7 (b) using the 2012 fixed daily drainage rate of 0.6 mm d^{-1}. Again, the model reproduces the measured daily depths accurately ($r^2 = 0.996$ for the 2013 drying period) and so a slow drainage rate of 0.6 mm d^{-1} was used in the wetland water balance model when depths were below 65 cms (before bund removal) and 54 cms (after bund removal).

4. Results

4.1. Water balance model performance

The overall performance of the water balance model is shown in Fig. 8 for the ten years 2013 to 2022. Daily modeled depths are generally similar to measured depths, with the annual mean root mean square error (RMSE) between the model and measured depths ranging from 7.2 cms (2013) to 30.7 cms (2022), Table 1. Values of the RMSE were higher in the years where there was no further optimization of the run in coefficient C that was set in 2013 and 2014. The largest difference between the model and measured values occurred in the wettest year (2019; $P =$ 2796 mm), where modelled values were well below measured depths for much of the year (March to July). This could have been due to the modelled drainage being greater than actual drainage which may have been impeded by thick weeds choking the wetland outlet in this year. Conversely, in the driest year (2015; annual $P = 1019$ mm) the model overestimated measured depths (RMSE = 9.2 cms), and this may have been due to the model overestimating run in to the wetland under these very dry conditions.

Over the entire 10 years of simulation the RMSE was 12.7 cms, indicating that the model can predict the wetland depth with reasonable

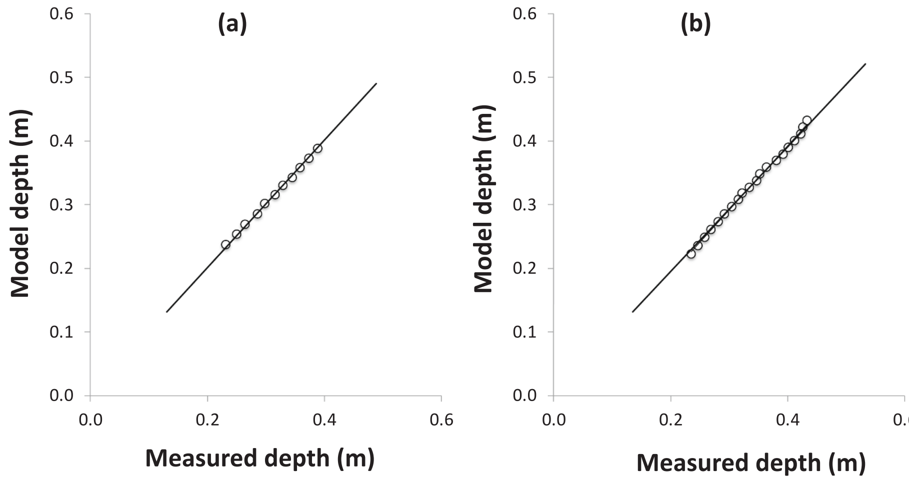

Fig. 7. Modeled and measured depth changes during rain and run-in free periods of slow drainage from the wetland; (a) 20/11/2012 to 10/12/2012; y = 1.00x, r^2 = 0.997 (b) 17/08/2013 to 25/09/2013.

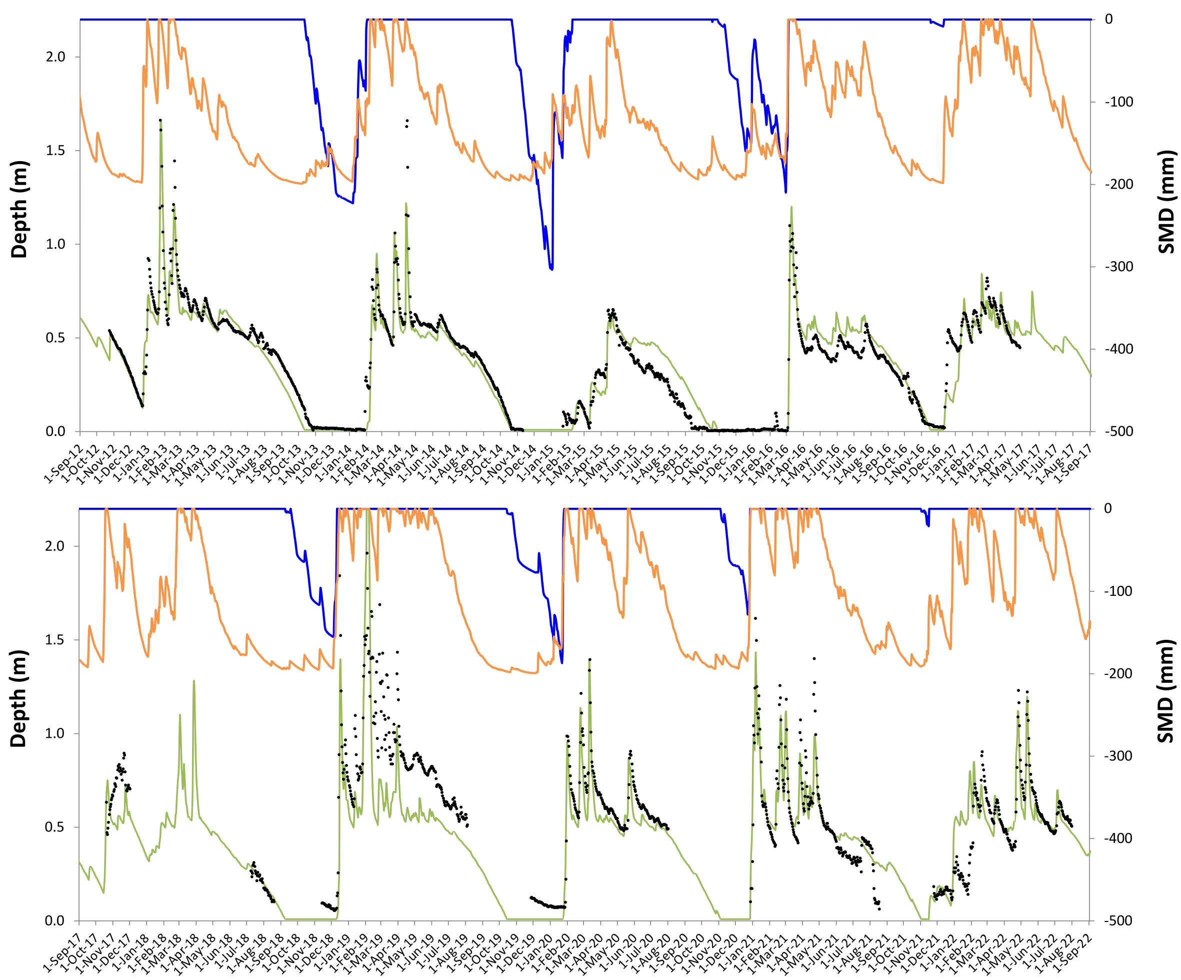

Fig. 8. Comparison of modeled (green) and measured (black) wetland depth over ten years from October 2012. The catchment soil moisture deficit (SMD_c) is shown in orange and the soil moisture deficit when there is no water in the wetland (SMD_w) is shown in blue.

accuracy most of the time.

The model also reproduces the peak depths that occur during the wet seasons fairly well (Fig. 8), which means it can be used in conjunction with topographic data to calculate the maximum area and volume of water in the wetland. The occurrence and duration of the periods of low or zero depth are also reasonably well estimated (to ~ ± 16 %) by the model and these are important in identifying when the conditions in the wetland pose a risk to fish and other aquatic species. This is discussed further in the section on ecological risks.

Table 1
The annual root mean square error (RMSE) between the modelled and measured daily depths for the 10 years studied. Also shown is the annual rainfall in each year.

Year	Rainfall (mm)	Mean depth (cms)	RSME (cms)	Optimized
2013	1263	46	7.2	Yes
2014	1617	42	8.2	Yes
2015	1019	19	9.2	No
2016	1552	32	9.5	No
2017	1559	63	12.5	No
2018	2012	31	12.2	No
2019	2796	78	30.7	No
2020	2083	57	10.3	No
2021	2541	51	9.9	No
2022	2037	51	17.0	No
10 year mean		47	12.7	

4.2. Annual water balance

The gains and losses of water from the wetland over the ten years 2013 to 2022 are shown in Fig. 9. The greatest input of water was from run in, with over twice as much water entering the wetland as run in compared with direct rainfall input. In the wettest year, (2019) run in was nearly three times the rainfall input, but only just exceeded the rainfall input in the dry 2015 year. The biggest loss of water from the wetland was via drainage, averaging 74 % of the total loss in all 10 years. However, drainage losses were highly variable, ranging from 94 % of the total water loss in the wettest year (2019), to only 14 % in the driest year (2015). Open water evaporation in the driest year was 53 % of the total water loss, with a further 33 % lost as evaporation from the soil in the wetland (when the depth was zero).

Over the entire 10-year period, evaporation from the open water surface made up ~ 22 % of the loss in these years and evaporation from the soil in the wetland (when the depth is zero) was only about 4 %. Clearly the prevailing weather (mainly rainfall) has a major effect on the wetland water balance. The model developed here can be used with readily available weather data (from SILO) to calculate the annual cycle in wetland depth over many years to identify the frequency and duration of periods when depths are low enough to present ecological risks.

5. Discussion

All of the surface and groundwater inputs and outputs in wetlands are rarely measured and sometimes too difficult to monitor, if there are multiple and/or diffuse fluxes (e.g. overland flow). This paper shows how to derive a complete water balance for a wetland from depth (and daily weather) measurements alone. The key steps for the general application of the wetland water balance model are as follows:

1. Determine the daily rate at which depth decreases (δd) from periods when there is little or no rainfall.
2. Calculate the daily evaporation rate (E_w) using an appropriate free water evaporation formula.
3. Estimate the daily drainage rate from the wetland by subtracting E_w from the values of δd obtained in 1 above. The drainage rate (D_w) can be plotted as a function of depth and the relationship obtained used to calculate D_w on all days.
4. Estimate of the amount of water entering the wetland as run in (R_{in}) using a simple runoff coefficient (C) model. The value of R_{in} needs to be adjusted according to the catchment soil moisture deficit (SMD_c) and if the wetland dries out, the wetland soil moisture deficit (SMD_w).
5. Daily values of R_{in} can be calculated using an exponential decay function to account for R_{in} continuing for a number of days after rainfall.
6. The value of C for any given wetland is determined by iteratively minimizing the root mean square error (RMSE) between the modelled and measured daily depths. A similar RMSE minimization is performed to obtain the exponential decay function coefficient (b).
7. Modelled changes in wetland depth for any past or future periods can then be calculated from Eq. (1) using readily available weather data.

The above steps can be applied in any wetland and the drainage rate function, evaporation rate and run in (via the value of C) will be specific to that wetland. Note that value of α in the soil evaporation formula (3.3) and the crop coefficient thresholds in Fig. 2 (100 and 200 mm) are specific to a site with a sandy soil and may need to be adjusted for locations with different soil characteristics.

When the above model was applied to the Mungalla wetland it showed that the largest input of water is from run in from the surrounding catchment and the greatest loss of water occurs as drainage. It

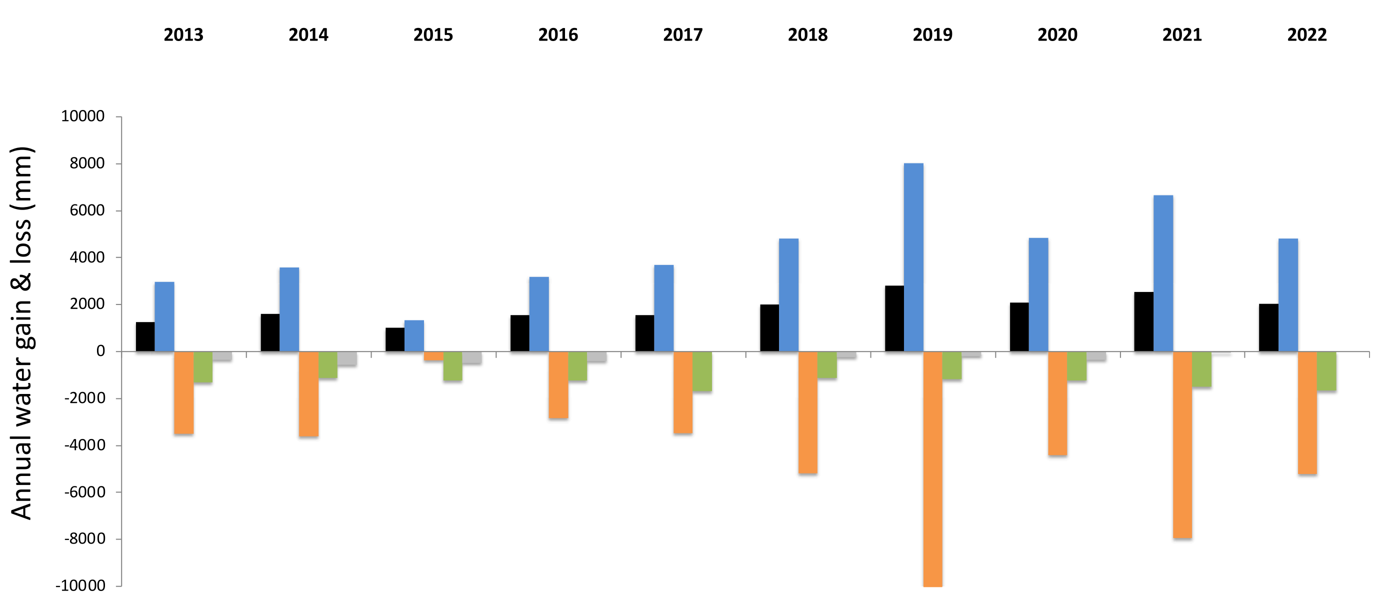

Fig. 9. The annual water balance of the wetland for 2013 to 2022. Water gains as direct rainfall (black) and run in (blue) are shown along with losses due to drainage (orange), water evaporation (green) and soil evaporation (grey).

is important to take into account the dryness of the surrounding catchment (SMD_c) as this has a major effect on how much run off is generated for any given rainfall event. If the wetland dries out completely at any time a moisture deficit (SMD_w) can also build up in the wetland soil. When subsequent rainfall and run in occur, this deficit must be replenished before the wetland water level can rise.

Drainage from the wetland occurs in two phases; rapid, when the wetland depth is deep and water can flow over the earth bund and slow when the water depth is shallow and the main loss mechanism is via groundwater seepage. Rapid drainage is the dominant component, but the slow drainage rate is important for estimating when depths are shallow and approach zero. In the Mungalla wetland drainage was much greater after the gap was created in the earth bund, for example, for the same wetland depth of 1 m, drainage after the gap was formed was 1.8 times that when the bund was intact. It is also possible that thick weed growth above and below the earth bund gap could impede drainage (Waltham et al., 2023b), but it usually remained greater than before the gap was created.

In the Mungalla wetland annual drainage was the greatest loss mechanism, but it was highly variable (14 to 94 % of the total loss) depending on the rainfall. The groundwater seepage component of drainage was generally small (0.56 mm day^{-1}) and independent of depths below 65 cms. If groundwater seepage was independent of wetland depths above this, we can estimate that it generally constituted between 2 and 7 % of total drainage. However, in the very dry year of 2015 groundwater seepage dominated drainage at 62 %. Evaporation from the wetland water was the next largest loss (averaging ~ 22 % of the total loss), and could even exceed drainage in dry years. In contrast, evaporation from the wetland soil when water was absent was small (only about 4 %), but as mentioned above was an important determinant of the period of zero water depth in the wetland.

The optimized water balance model was generally able to estimate the Mungalla wetland depths to around ± 12.7 cms and was able to simulate peak depths and low/zero depths fairly well. A similar estimation accuracy, ± 12.1 cms was obtained when the water balance model was applied in a 2.5 ha constructed wetland in the dry tropics (annual rainfall 1738 mm) near Mackay (Wallace et al., 2022). The absolute depth error was greater in the 10 ha Babinda constructed wetland at ± 27.1 cms, where the annual rainfall is 4287 mm (Wallace and Waltham, 2021). The greater depth uncertainty in the Babinda wetland was due to the occurrence of frequent large flood pulses which caused very rapid increases and decreases in the wetland depth, which are more difficult to model. This is also reflected in the Mungalla wetland study, where greater uncertainty in depth estimation occurred in wetter years. Overall, given the relative simplicity of the methods for deriving the wetland water balance the modelled estimates of depth are reasonably good and suggest that the model could be applied in many more wetlands both in Australia and elsewhere. To date the model has been used successfully in wetlands ranging from 2.5 to 160 ha, but applications to larger wetlands have yet to be tested.

The above wetland water balance model can provide useful insights into ecological impacts and nutrient and sediment removal under both current and future climate conditions. For example, fish and other aquatic biota may experience thermal stress or even death if water depths in a wetland fall below certain levels. For example, Waltham et al., (2020) reported that when depths fell below ~ 40 cms in the Mungalla wetland, water temperatures could exceed the tolerance thresholds of acute exposure to aquatic species of many tropical fish (Burrows and Butler 2012) and crustaceans (Waltham, 2018), which presents a major challenge for the services that coastal wetlands, in terms of providing important habitat for fisheries, offer in these areas (Barbier et al., 2011; Elliott & Whitfield 2011). Wetlands can also have significant periods when there is no water, when clearly fish and other aquatic biota cannot survive unless they can migrate to other waterway locations in the floodplain network. When water returns to a wetland after it was empty there may also be a period when the water becomes quite acidic when the pH drops to ~ 3.5 for several weeks (Waltham et al., 2020). These drops in pH are associated with reflooding of dried out acid sulphate soils, occurring at the end of the dry season and they can cause fish kills (Pearson et al., 2021; Waterhouse et al., 2016). There are therefore three periods when there can be significant risks to aquatic biota: (1) when depths are low, (2) when the depth is zero and (3) when water returns to a wetland. All three of these risk periods can be predicted using the wetland water balance model described here using readily available local weather data. Indeed, by using historic weather data, the water balance model described here could be used to construct a time series of how these risk periods have varied over time, giving insights into how key ecological conditions in the wetland have evolved. Since the water balance model is driven by weather data, then if future climate change predictions are available, it will also be possible to predict how these risk periods might change in the future. Note that the timing and duration of these risk periods will be specific to each wetland according to its size (area and volume) and location in the landscape (that determine its hydrological regime). Understanding this exposure risk to aquatic species could be an additional tool for managers when planning and implementing restoration activities in coastal wetlands of this nature.

Our water balance modelling approach can also be used in conjunction with a nutrient balance model to estimate the removal of nitrogen and sediment from wetlands. A recent example of this has been reported by Wallace et al., (2022) for a constructed wetland at Bakers Creek, near Mackay, north Queensland. The nutrient balance model requires the concentrations of dissolved and particulate nitrogen entering the wetland, along with estimates of the denitrification rate within the wetland. Even with incomplete data of this type, the above authors were able to estimate that this 2.5 ha wetland removed 83 ka N ha^{-1} year^{-1} (37 % of its nitrogen load). This then allows the area of wetlands of this type that are required to help meet the nitrogen and sediment reduction targets for the catchment (Pioneer) to be calculated. As climate change may alter the future rainfall and temperature, this type of combined water and nutrient balance modelling can also be used to examine the potential impact of future weather scenarios on the nutrient and sediment removal of wetlands.

Different wetlands in different locations will remove varying amounts of nitrogen and sediment (Mitsch et al., 2012). To assess how this will vary within and between catchments along the GBR coast, it will be necessary to nest wetland water and nutrient balance models within catchment scale hydrological models (Waltham, 2023a). These latter models use rainfall-runoff algorithims and stream water routing routines to quantify water movement across a catchment (Brodie et al., 2009; Liu et al., 2021). In principle, they can provide a time series of flow at specified points within a catchment. This could provide the input to a nested wetland water balance model; the final parameters needed are concurrent time series of nitrogen and sediment concentrations in the wetland inflow. There is the possibility of developing existing water quality models to estimate the required nitrogen and sediment concentrations. If this can be achieved, it should then be possible to examine how altering the characteristics of a wetland (type, size, vegetation content etc.) and its location within a catchment flow network affect its nutrient and sediment removal capability.

6. Conclusions

The water balance model described here has the advantage that it can provide an estimate of the main inflows and outflows to and from a wetland without the need for their direct measurement. The only daily variables required are weather data and the wetland depth, which is easy and inexpensive to measure with deployment of simple and relatively inexpensive high frequency loggers in the wetland. The model has been applied here and in two other Australian wetlands (Wallace and Waltham, 2021; Wallace et al., 2022) with very different rainfall regimes and has the potential to be applied in many more wetland locations. The current application in Mungalla uses Australia's longest wetland data set (10 years) from which it has been possible to demonstrate that inflows and outflows are highly variable between years. This illustrates the highly dynamic nature of these wetlands which has important implications for their ecological condition.

However, such a simple method may have significant uncertainty in the estimated inflows and outflows, and it only gives a crude estimate of groundwater fluxes. Nevertheless, in the absence of direct measurement of wetland inflows, outflows and groundwater fluxes the method provides a very valuable first estimate that can be used in a number of subsequent applications. The uncertainty in the above simple water balance method could be evaluated in wetlands that are fully monitored for surface inflows and outflows along with estimates of any significant groundwater fluxes.

Another novel application of this water balance model is that it can determine aquatic risk periods and how these have changed historically, thereby giving insights into the evolution of ecological conditions in a wetland. Future ecological conditions may also be interpreted from estimates of how these aquatic risks might change under a future climate. In addition, this model can also be combined with a nutrient balance model to estimate the nitrogen and sediment removal of different wetlands in different locations. There is also the potential for nesting such water and nutrient models within broader catchment scale runoff and water quality models. This approach will be very valuable in helping evaluate the type and location of wetlands that can make the best contribution to GBR load reductions, particularly under increasing interest in the application of treatment wetlands in meeting GBR water quality targets and payment for ecosystem services (Waltham et al., 2021).

CRediT authorship contribution statement

J. Wallace: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Conceptualization. **M. Nicholas:** Writing – review & editing, Project administration, Funding acquisition, Data curation, Conceptualization. **A. Grice:** Supervision, Project administration, Investigation, Funding acquisition, Data curation, Conceptualization. **N.J. Waltham:** Writing – review & editing, Project administration, Investigation, Funding acquisition, Data curation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The authors do not have permission to share data.

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References

Abbott, B.N., Wallace, J., Nicholas, D.M., Karim, F. and Waltham, N.J. 2020. Bund removal to re-establish tidal flow, remove aquatic weeds and restore coastal wetland services – North Queensland, Australia. *PLOS ONE*. doi: 10.1371/journal.pone.0217531.

Adame, M.F., Arthington, A.H., Waltham, N., Hasan, S., Selles, A., Ronan, M., 2019a. Managing threats and restoring wetlands within catchments of the Great Barrier Reef, Australia. Aquat. Conserv. Mar. Freshwat. Ecosyst. 29 (5), 829–839.

Adame, M.F., Franklin, H., Waltham, N.J., Rodriguez, S., Kavehei, E., Turschwell, M.P., Balcombe, S.R., Kaniewska, P., Burford, M.A., Ronan, M., 2019b. Nitrogen removal by tropical floodplain wetlands through denitrification. Mar. Freshw. Res. 70 (11).

Adame, M.F., Waltham, N.J., Iram, N., Farahani, B.S., Salinas, C., Burford, M., Ronan, M., 2021. Denitrification within the sediments and epiphyton of tropical macrophyte stands. Inland Waters 11 (3), 257–266.

Allen, R. G., Pereira, L. S., Raes, D., & Smith, M. (1998). FAO Irrigation and drainage paper No. 56. *Rome: Food and Agriculture Organization of the United Nations*, *56*(97), pp156.

Arthington, A.H., Godfrey, P.C., Pearson, R.G., Karim, F., Wallace, J., 2015. Biodiversity values of remnant freshwater floodplain lagoons in agricultural catchments: evidence for fish of the Wet Tropics bioregion, northern Australia. Aquat. Conserv. Mar. Freshwat. Ecosyst. 25, 336–352. https://doi.org/10.1002/aqc.2489.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81 (2), 169–193.

Brodie, J., Lewis, S., Bainbridge, Z., Mitchell, A., Waterhouse, J., Kroon, F., 2009. Target setting for pollutant discharge management of rivers in the Great Barrier Reef catchment area. Mar. Freshw. Res. 60 (11), 1141–1149.

Burk, L., Dalgleish, N., 2013. Estimating plant available water capacity. Grains Research and Development Corporation, ACT, Australia, p. 24. ISBN 978-1-875477-84-5.

Burrows, D. and Butler, B. 2012. Preliminary studies of temperature regimes and temperature tolerance of aquatic fauna in freshwater habitats of northern Australia. Centre for Tropical Water and Aquatic Ecosystem Research, Report 12/01 (February 2012), pp46.

Butler, B., Loong, D., Burows, D. and Morgan, G. 2009. Aquatic ecology assessment of Mungalla wetlands: May to August 2009. A Report to CSIRO Prepared for the Mungalla Coastal Habitat Management Project. ACTFR Report No, 09/38 December 2009. 92pp.

Canning, A.D., Smart, J.C., Dyke, J., Curwen, G., Hasan, S., Waltham, N.J., 2023. Constructed wetlands suitability for sugarcane profitability, freshwater biodiversity and ecosystem services. Environ. Manag. 71 (2), 304–320.

Canning, A.D., Waltham, N.J., 2021. Ecological impact assessment of climate change and habitat loss on wetland vertebrate assemblages of the Great Barrier Reef catchment and the influence of survey bias. Ecol. Evol. 11 (10), 5244–5254.

Critchley, W. and Siegert, K. 1991. Water Harvesting: A Manual for the Design and Construction of Water Harvesting Schemes for Plant Production. Food and Agriculture Organization of the United Nations - Rome, 1991.

Davis, A.M., Pearson, R.G., Brodie, J.E., Butler, B., 2016. Review and conceptual models of agricultural impacts and water quality in waterways of the Great Barrier Reef catchment area. Mar. Freshw. Res. 68 (1), 1–19.

Elliott, M., Whitfield, A.K., 2011. Challenging paradigms in estuarine ecology and management. Estuar. Coast. Shelf Sci. 94 (4), 306–314.

Flint, N., Crossland, M.R., Pearson, R.G., 2015. Sublethal effects of fluctuating hypoxia on juvenile tropical Australian freshwater fish. Mar. Freshw. Res. 66, 293–304. https://doi.org/10.1071/MF14120.

Furnas, 2003 Catchments and Corals- Terrestrial runoff to the Great Barrier Reef. AIMS/CRC Reef. ISBN 0 642 55041 7.

Great Barrier Reef Marine Park Authority, 2014. Great Barrier Reef Region Strategic Assessment: Strategic assessment report. GBRMPA, Townsville, p. 684.

Grice, A.C., Cassady, J., Nicholas, D.M., 2012. Indigenous and non-Indigenous knowledge and values combine to support management of Nywaigi lands in the Queensland coastal tropics. Ecol. Manage. Restor. 13, 93–97.

Jeffrey, S.J., Carter, J.O., Moodie, K.B., Beswick, A.R., 2001. Using spatial interpolation to construct a comprehensive archive of Australian climate data. Environ Model Softw. 16, 309–330.

Karim, F., Wallace, J., Abbott, B., Waltham, N.J., Marvanek, S., Nicholas, M., 2021. Modelling the removal of an earth bund to maximise seawater ingress into a coastal wetland in north Queensland, Australia. Estuar. Coastal Shelf. Science:107626.

Kavehei, E., Roberts, M.E., Cadier, C., Griffiths, M., Argent, S., Hamilton, D.P., Lu, J., Bayley, M., Adame, M.F., 2021. Nitrogen processing by treatment wetlands in a tropical catchment dominated by agricultural landuse. Mar. Pollut. Bull. 172, 112800.

Liu, S., Ryu, D., Webb, J.A., Lintern, A., Guo, D., Waters, D., Western, A.W., 2021. A multi-model approach to assessing the impacts of catchment characteristics on spatial water quality in the Great Barrier Reef catchments. Environ. Pollut. 288, 117337.

McJannet, D.L., Cook, F.J., Burn, S., 2013. Comparison of techniques for estimating evaporation from an irrigation water storage. Water Resour. Res. 49, 1415–1428. https://doi.org/10.1002/wrcr.20125.

McJannet, D. L., Webster, I. T., Stenson, M. P. and Sherman, B. S. 2008. Estimating open water evaporation for the Murray Darling Basin. A report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project. DOI: 10.1111/J.1095-8649.2008.01959.X.

McJannet, D., Wallace, J., Keen, R., Hawdon, A., Kemei, J., 2012a. The filtering capacity of a tropical riverine wetland: I water balance. Hydrol. Process. 26, 40–52.

McJannet, D., Wallace, J., Keen, R., Hawdon, A., Kemei, J., 2012b. The filtering capacity of a tropical riverine wetland: II sediment and nutrient Balances. Hydrol. Process. 26, 53–72.

Mitsch, W.J., Zhang, L., Stefanik, K.C., Nahlik, A.M., Anderson, C.J., Bernal, B., Hernandez, M., Song, K., 2012. Creating wetlands: primary succession, water quality changes, and self-design over 15 years. Bioscience 62 (3), 237–250.

Pearson, R.G., Godfrey, P.C., Arthington, A.H., Wallace, J., Karim, F., Ellison, M., 2013. Biophysical status of remnant freshwater floodplain lagoons in the Great Barrier Reef catchment: a challenge for assessment and monitoring. Mar. Freshw. Res. 64, 208–222. https://doi.org/10.1071/MF12251.

Pearson, R.G., Connolly, N.M., Davis, A.M., Brodie, J.E., 2021. Fresh waters and estuaries of the Great Barrier Reef catchment: effects and management of anthropogenic disturbance on biodiversity, ecology and connectivity. Mar. Pollut. Bull. 166, 112194.

Perna, C.N., Cappo, M., Pusey, B.J., Burrows, D.W., Pearson, R.G., 2012. Removal of aquatic weeds greatly enhances fish community richness and diversity: an example from the Burdekin River floodplain, tropical Australia. River Res. Appl. 28, 1093–1104. https://doi.org/10.1002/rra.1505.

Pilgrim, D.H. and Crodery, I. 1993. Flood runoff. In: Handbook of Hydrology (Ed. D.R. Maidment). pp9.1 – 9.42. McGraw-Hill, New York.

Ritchie, J.T., 1972. Model for predicting evaporation from a row crop with incomplete cover. Water Resour. Res. 8, 1204–1213.

Sheaves, M., Brookes, J., Coles, R., Freckelton, M., Groves, P., Johnston, R., Winberg, P., 2014. Repair and revitalization of Australia's tropical estuaries and coastal wetlands: opportunities and constraints for the reinstatement of lost function and productivity. Mar. Policy 47, 23–38. https://doi.org/10.1016/j.marpol.2014.01.024.

Shuttleworth, W.J. 1993. Evaporation. In: Handbook of Hydrology (D.R. Maidment) pp 4.1 - 4.53.

Twomey, A.J., Nunez, K., Carr, J.A., Crooks, S., Friess, D.A., Glamore, W., Orr, M., Reef, R., Rogers, K., Waltham, N.J., Lovelock, C.E., 2024. Planning hydrological restoration of coastal wetlands: key model considerations and solutions. Sci. Total Environ., 169881

Wallace, J., Bueno, C., Waltham, N.J., 2022. Modelling the removal of nitrogen and sediment by a constructed wetland system in north Queensland, Australia. Ecol. Eng. 184, 106767.

Wallace, J., Adame, M.F., Waltham N.J. (2020). A constructed wetland near Babinda, north Queensland: a case study of potential water quality benefits in an agricultural tropical catchment (2020). Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, Cairns (67pp.).

Wallace, J., Waltham, N.J., 2021. On the potential for improving water quality entering the Great Barrie Reef using constructed wetlands. Mar. Pollut. Bull. 170 (2021), 112627.

Waltham, N.J., 2018. Acute thermal effects in an inland freshwater crab *Austrothelphusa transversa* (von Martens, 1868) occupying seasonal, tropical rivers. J. Crustac. Biol. 38, 497–503.

Waltham, N.J. and Sheaves, M. 2015. Waltham, N. J., Adame, M. F., Karim, F., Abbott, B., Wallace, J. (2020) Bund wall removal increases saltwater intrusion and controls invasive aquatic weeds on a coastal floodplain. Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, Cairns (83pp.).

Waltham, N.J., Fixler, S., 2017. Aerial herbicide spray to control invasive water hyacinth (Eichhornia crassipes): water quality concerns fronting fish occupying a tropical floodplain wetland. Trop. Conserv. Sci. 10, 1940082917741592.

Waltham, N.J., Burrows, D., Wegscheidl, C., Buelow, C., Ronan, M., Connolly, N., Groves, P., Marie-Audas, D., Creighton, C., Sheaves, M., 2019. Lost floodplain wetland environments and efforts to restore connectivity, habitat, and water quality settings on the Great Barrier Reef. Front. Mar. Sci. 6, 71.

Waltham, N. J., Adame, M. F., Karim, F., Abbott, B., Wallace, J. (2020) Bund wall removal increases saltwater intrusion and controls invasive aquatic weeds on a coastal floodplain. Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, Cairns (83pp.).

Waltham N, Karim F and Wallace J, 2023b. Modelling bund wall modification and vegetation removal to maximise tidal water ingress: Mungalla wetland complex. Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication 22_48, James Cook University, Townsville, Australia, 21pp.

Waltham, N.J., Wegscheidl, C., Volders, A., Smart, J.C., Hasan, S., Lédée, E., Waterhouse, J., 2021. Land use conversion to improve water quality in high DIN risk, low-lying sugarcane areas of the Great Barrier Reef catchments. Mar. Pollut. Bull. 167, 112373.

Waltham, N. J., Motson, k. and Molinari, B. 2023a. Scientific Consensus Statement: Synthesis of the evidence. *4.7 What is the efficacy of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality (nutrients, fine sediments and pesticides)? 51pp.*

Waterhouse, J., Brodie, J., Lewis, S., Audas, D.M., 2016. Land-sea connectivity, ecohydrology and holistic management of the Great Barrier Reef and its catchments: time for a change. Ecohydrol. Hydrobiol. 16 (1), 45–57.