

Mangrove carbon assessment tool: Model validation and assessment of mangroves in southern USA and Mexico

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ABSTRACT

While mangroves are widely recognized as a significant carbon (C) reservoir and the valued ecosystem services are inextricably linked to the C stocks and fluxes. Modeling tools haven't been available to simulate C dynamics in mangroves to inform assessments, Monitoring, Reporting and Verification for REDD+, or management and restoration prescriptions. The process-based model MCAT-DNDC (Mangrove-Carbon-Assessment-Tool-DeNitrification-DeComposition) was validated using measurements from sites in Quintana Roo, Mexico and Florida, USA. The validated model was then applied to model C sequestration in mangroves sites in Texas, Louisiana and Florida that had measured data for comparison. The model validation against aboveground biomass (AGB) showed that the simulation provided good agreement with observations with a proper slope (1.06) and small intercept (1.32 Mg ha⁻¹, about 1.4% of observed mean); the model performance efficiency for assessing AGB was high (R² = 0.99). Among ten C pools and fluxes validated using data from the Everglades National Park, eight components were in good agreement with the observations, and two were within the range of observation; demonstrating effective model performance (R² > 0.95). The metrics from the model validation showed that MCAT-DNDC can be used to estimate C sequestration in mangroves within the coastal areas along Gulf of Mexico and Mexican Caribbean with good model performance. Simulated C dynamics for plots in Texas, Louisiana and Florida showed that the relationship between above-ground biomass and stand age was non-linear, and that gross and net primary productivity increased logarithmically with stand age. The differences in C components among the sites exhibited the effects of ecological drivers on C sequestration in mangroves. Simulations also demonstrated that the model may be useful in considering the effect of forest management on C sequestration. The model appears to be stable and sufficiently robust to warrant further testing with additional data and across a variety of sites.

1. Introduction

Mangroves are one of the most carbon-rich forest ecosystems in the tropics due largely to the carbon (C) accumulation in soils (Donato et al., 2011); the C stocks are the basis of the many valued ecosystem services provided by mangroves to coastal and nearshore areas (Bouillon et al., 2008; Alongi, 2009; Rahman et al., 2015). As a result of the large ecosystem C stocks in mangroves, they're considered significant to the global C cycle, despite a relatively small area globally (Mudiyarso et al., 2015; Attwood et al., 2017). Because of differences in ecological drivers, carbon stocks in mangrove ecosystems vary widely, corresponding with a wide variety of mangrove types, and climatic and hydrogeological conditions (Kauffman et al., 2011; Alongi, 2014;

Rahman et al., 2015; Sanders et al., 2016; Estrada and Soares, 2017).

Mangroves are also threatened as a result of deforestation, land use conversion, and natural disturbances, which in turn can result in significant losses of both soil and vegetation C stocks (Attwood et al., 2017). The mangrove landscape is not static, and newly formed stands can be sites of C sequestration. Correspondingly, mangrove expansion, deforestation or other disturbance regimes can impact C storage in biomass as well as soils (Perez et al., 2017). Given the linkage between C stocks and ecosystem service it would be helpful to have a model that provided capabilities for assessing C dynamics in mangroves. This is especially true for REDD+ (Reducing Emissions from Deforestation and forest Degradation, plus conserving forests and promoting sustainable forest management) and other payment for ecosystem services

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initiatives that depend on accurate assessments of C stocks and fluxes. An assessment tool is also needed for considering the potential effects of forest management regimes on C stocks. Models can also help elucidate natural processes that regulate C dynamics providing a basis for both experimentation and observation to support basic and applied assessments, as well as the development of management practices.

A mechanistic and spatially-explicit model, MCAT-DNDC [Mangrove-Carbon-Assessment-Tool (MCAT), DeNitrification-DeComposition (DNDC)], has been developed from expert knowledge, measurements and observations to estimate C, nitrogen (N) and phosphorous (P) dynamics in mangroves, in support of applied assessments including MRV (Monitoring, Reporting and Verification) for REDD+. It is also intended to provide a basis for simulating C cycle response in mangroves to climate change, sea level rise and other natural and anthropogenic disturbances.

MCAT-DNDC was developed by integrating biogeochemical processes of Forest-DNDC (Li et al., 2000), which accommodated freshwater wetland biogeochemistry, with new provisions for C, N and P processing to accommodate changes in biogeochemical reactions mediated thru the marine hydrology and sediments. This new model simulates (1) C sequestration in mangroves, including gross and net primary productivity (GPP and NPP, respectfully), above- and below-ground biomass (AGB and BGB) allocation based on energy balance in the ecosystems and plant nutrient demand, (2) organic matter decomposition processes (3) C fluxes in terms of gas emissions (e.g., CO₂, CH₄, N₂O), from the soil surface and leaching [e.g., DIC (dissolved inorganic C), DOC (dissolved organic C), and POC (particulate organic C)] that are influenced by changes in ecological and hydrological drivers, and (4) C accumulation and turnover within the soil. Important regulators of the C-cycle include nutrients, especially P, salt stress, disturbance regimes and changes in ecological drivers (see the detailed modeling approach in Dai et al. 2018). The model is spatially-explicit, and can be used to simulate C, N and P dynamics in mangroves within a designated area, ranging from a single profile of plant-soil conditions to a watershed or larger region using GIS polygons to provide spatially specific vegetation, soil, climate, hydrology (including tides) and topographic data. The details about the model are provided by Dai et al. (2018).

The objectives of this study were twofold: (1) to validate the process-based mangrove carbon assessment tool MCAT-DNDC using measurements from field-based studies, and (2) demonstrate the applicability of the model for simulating C dynamics in a variety of mangrove stands. The model validation was based on comparisons with measured data from process-level studies conducted on eight sites in Florida and Mexico: the Everglades National Park (ENP) site in southern Florida of USA (Castaneda-Moya et al., 2013) that is well-studied (Twilley, 1985; Chen and Twilley, 1999; Romigh et al., 2006; Barr et al. 2010, 2013; Castaneda-Moya et al., 2013), and seven UNESCO World Heritage sites within Mexican Caribbean in Quintana Roo (QR) (Adame et al., 2013). Model applicability was assessed using data from three sites in coastal Texas (TX), Louisiana (LA) and mid-west Florida (FL) of USA. Each of the three sites had nine replicate plots, which provided useful differences in ecological conditions for assessing C sequestration.

2. Methods

2.1. Study sites

A total of 35 research plots (Fig. 1 and Tables 1 and 2) were available from sites within the Caribbean basin (Quintana Roo, Mexico) and coastal Southern USA (Texas, Louisiana and Florida) were used for model validation and assessment. The south Florida site (ENP) is located at 25.3646° N, 81.0779° W in the Everglades National Park, and the seven plots from the UNESCO World Heritage site in Quintana Roo (QR) are distributed along the Caribbean coast from 19.4867° to 19.8612° N, 87.495°–87.7004° W. Eight plots from these two sites were used for model validation (see Table 1). The other 27 plots, located in

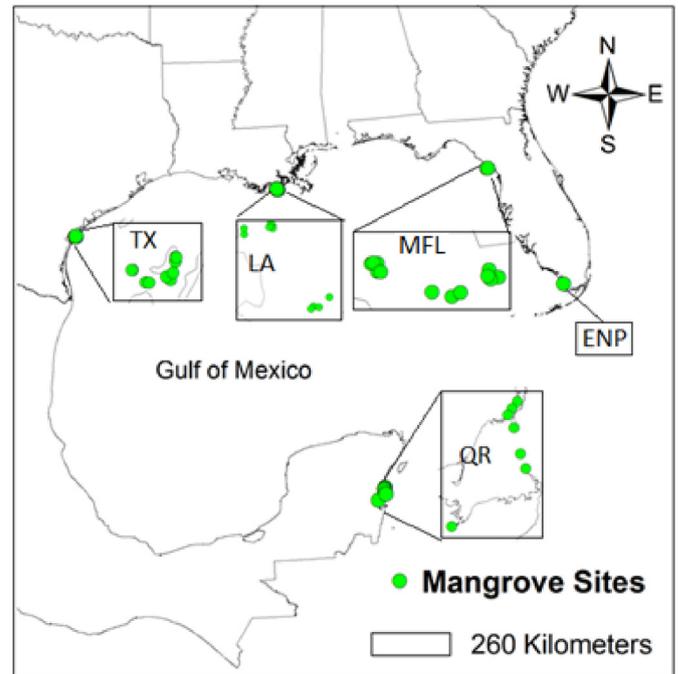


Fig. 1. Location of study sites and plots. The Everglades National Park (ENP) and plots in Quintana Roo (QR), Mexico were used for the model validation; while the plots in Florida (FL), Louisiana (LA) and Texas (TX) were used for model applications.

FL, LA and TX (Osland, 2014) were used to assess C spatiotemporal dynamics (Table 2). Because of variation in environmental conditions, there were considerable differences in the canopy stature among these plots, ranging from dwarf (< 1 m in height) to tall mangroves (> 10 m in height).

The mangrove species common to the region are represented in the data used from ENP and QR for validation (Table 1), in contrast only *Avicennia* was only species in the sites in the Gulf of Mexico sites in FL, LA and TX (Table 2). Physiochemical soil properties are substantially different among the sites, including differences in pore water salinity, and soil P and organic C (see Tables 1 and 2). Climatic conditions varied considerably among these sites. Mean air temperature in the 35-year period from 1980 to 2014 ranged from 20.9 at the plots in Louisiana to 26.4 °C in Quintana Roo, and the mean annual precipitation varied between 821 mm at TX and 1611 mm at LA (Tables 1 and 2). The high temperature over the 35-year period was 45 °C at QR and approximately 38 °C at the other sites. Freezing temperatures (< 0 °C), which adversely affects mangroves (Osland et al., 2013), can occur in LA, TX, and FL, but not in ENP and QR, based on climate data for the 35-year period from 1980 to 2014.

2.2. Climate, soil and vegetation data

Climate data for all sites were downloaded from Daymet database (Thornton et al., 2012; <http://daymet.ornl.gov/index.html>), including daily minimum and maximum temperature and daily precipitation for a time period from 1980 to 2014. Vegetation and soil data used for the model validation were from Castaneda-Moya et al. (2013) for the ENP plot in Florida, and Adame et al. (2013) for the seven plots in QR. To assess C sequestration in the 27 plots in Florida, Louisiana and Texas (Fig. 1), vegetation and soil data were obtained from inventories conducted by Osland et al. (U.S. Geological Survey, Lafayette, Louisiana 70506 USA), downloaded from USGS database (<http://databasin.org/datasets/>).

Table 1

Eco-environmental characteristics of the plots from the Everglades National Park (ENP) and Quintana Roo, Mexico (QR1-7) that were used for the model validation.

Site	ENP	QR1	QR2	QR3	QR4	QR5	QR6	QR7
Latitude (degree)	25.3646	19.8612	19.8408	19.8218	19.4867	19.78	19.6957	19.6477
Longitude (degree)	−81.0779	−87.4612	−87.48	−87.495	−87.7004	−87.4789	−87.4659	−87.454
Temp ^S (°C)	24.5	25.9	25.9	25.9	26.3	26.0	26.0	26.1
Rainfall (mm)	1360	1231	1256	1284	1474	1341	1330	1555
Species [*]	A, L, R	R	R	R	L, R	R	R	R
Soil C [#] (%)	21.4	4.8	8.4	12.5	32.3	15.7	13.1	11.4
Soil N [#] (mg/g)	13.16	1.84	3.29	4.99	13.41	6.97	8.69	3.69
Soil P [#] (mg/g)	1.1	0.065	0.067	0.153	0.507	0.193	0.336	0.163
Salinity (ppt)	27.0	57.2	50.0 [@]	49.6	28.6	44.9	38.9	44.9 [@]
height ^α (m)	5.0–15	0.4–1.5	0.1–1.3	0.6–1.4	3.0–10.0	2.0–5.0	3.0–14.0	2.0–11
VOMV ^β	10 < SUP > Ω < /SUP >	AGB	AGB	AGB	AGB	AGB	AGB	AGB

^S: Temp, mean air temperature for a 35-year period from 1980 to 2014 in ENP and QR.^{*}: See details in [Castaneda-Moya et al. \(2013\)](#) for ENP in Florida, and [Adame et al. \(2013\)](#) for Mexican plots; A, *Avicennia germinans*, L, *Laguncularia racemosa*, R, *Rhizophora mangle*.[#]: soil C, soil organic carbon, unit, %; units for soil N and P, mg g^{−1}, the content of C, N and P is depth weighted average within 100 cm.[@]: salinity for these two plots are estimated based on the salinity of their neighbors; α: height, canopy height, meters.^β: VOMV, variables observed for the model validation.Ω: a total of 10 measured variables from ENP, including above ground biomass (AGB), above ground net primary productivity (ANPP), burial carbon (BC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), CH₄, particulate organic carbon (POC), leaf litter, total litter, reproductive parts from this plot were used for model validation; AGB from the QR plots was used for the model validation.

2.3. Water table data

Water table (WT) data is required for simulating C dynamics in wetlands. Unfortunately, WT observations from these study sites were not available for our simulation period, with the exception of ENP, which had data for daily WT in 2002. Accordingly, we developed a WT dataset based on the pattern of daily WT in 2002 at ENP ([Castaneda-Moya et al., 2013](#)), and this dataset was only used for model validation for ENP. To assess long-term mangrove C dynamics in ENP, and plots in QR, FL, LA and TX, the daily WT dynamics were estimated based on tidal calculations ([Schureman, 1941](#)) for these locations with the following assumptions: (1) tidal force was only influenced by the Moon

and Sun ([Longman, 1959](#)), and (2) the highest tides occurred at the time when the tidal force was the largest. The mean high tidal level (MHL) and mean low tidal level (MLL) for FL, LA and TX were downloaded from NOAA tidal database (https://tidesandcurrents.noaa.gov/tide_predictions.html). Estimated daily WT was averaged from the hourly WT based on tidal calculation ([Schureman, 1941](#)).

Precipitation may affect the WT, and it may influence salinity during and after rain events. We calculated the net daily precipitation added to the soil was:

$$\Delta P = P_r - PET \quad (1)$$

where ΔP is the net precipitation that can raise WT (cm d^{−1}) during

Table 2Eco-environmental characteristics of plots in Florida (FL1-9), Louisiana (LA1-9) and Texas (TX1-9) that were used to assess mangrove C dynamics ^{*}.

Site	Latitude (°)	Longitude (°)	Temp (°C)	Rainfall (mm)	Species	Soil C [#] (%)	Soil C ^S (%)	Salinity (ppt)	Cover (%)	Height (m) < SUP > Ω < /SUP >
FL1	29.1437	−83.0314	21.1	1560	A	14.31	10.90	30.67	80	4.3
FL2	29.1437	−83.0319	21.1	1560	A	17.43	12.94	28.83	65	2.8
FL3	29.1431	−83.0315	21.1	1560	A	14.56	13.28	29.93	80	2.8
FL4	29.1422	−83.0228	21.1	1560	A	12.32	9.73	29.87	90	2.2
FL5	29.1416	−83.0221	21.1	1560	A	11.49	8.93	35.97	70	4.8
FL6	29.1415	−83.0228	21.1	1560	A	13.81	9.37	52.30	80	3.5
FL7	29.1407	−83.0260	21.1	1560	A	11.43	8.83	38.63	70	5.3
FL8	29.1409	−83.0253	21.1	1560	A	15.83	11.27	36.37	80	5.0
FL9	29.1412	−83.0275	21.1	1560	A	11.29	10.67	30.00	80	6.5
LA1	29.1114	−90.1945	20.9	1614	A	4.52	3.62	44.77	85	2.3
LA2	29.1070	−90.2046	20.9	1617	A	3.79	2.03	50.43	80	N/A
LA3	29.1057	−90.2061	20.9	1617	A	4.65	3.53	47.27	95	2.7
LA4	29.1518	−90.2265	20.9	1611	A	5.30	5.90	48.10	80	1.6
LA5	29.1480	−90.2442	20.9	1611	A	4.27	5.49	31.77	60	1.5
LA6	29.1511	−90.2441	20.9	1612	A	3.66	5.55	53.57	65	N/A
LA7	29.1505	−90.2268	20.9	1611	A	5.38	4.47	32.37	90	1.7
LA8	29.1503	−90.2265	20.9	1611	A	6.22	4.57	47.93	85	1.6
LA9	29.1529	−90.2275	20.9	1611	A	5.68	4.62	43.00	75	N/A
TX1	27.8610	−97.0738	22.4	833	A	4.81	0.71	79.70	55	N/A
TX2	27.8627	−97.0559	22.4	838	A	1.50	0.42	57.20	30	N/A
TX3	27.8649	−97.0591	22.4	836	A	3.36	0.38	51.87	40	1.5
TX4	27.8752	−97.0525	22.4	840	A	4.16	0.99	47.67	90	2.1
TX5	27.8690	−97.0836	22.4	830	A	5.43	0.87	42.40	75	1.8
TX6	27.8689	−97.0843	22.4	830	A	1.97	0.38	61.27	35	N/A
TX7	27.8751	−97.0526	22.4	840	A	3.90	0.84	42.17	95	1.6
TX8	27.8748	−97.0525	22.4	840	A	3.72	0.65	37.53	85	1.9
TX9	27.8674	−97.0542	22.4	840	A	3.15	0.48	52.37	51	1.4

^{*}: Temp, mean air temperature for a 35-year period from 1980 to 2014; rainfall, precipitation in the same period; species A, *Avicennia germinans*; Soil C[#], soil C content within 0–5 cm in depth; Soil C^S, soil C content within 5–15 cm in depth; Cover, canopy cover rate; vegetation and soil data, obtained from USGS database (<http://datbasin.org/datasets/>); Ω: mean canopy height; N/A, data not available.

rain, and $\Delta P = 0$ if $P_r < PET$; P_r is daily precipitation (cm d^{-1}) and PET (cm d^{-1}) is daily potential evapotranspiration estimated using Hargreaves equation (Gavilan et al., 2006; Sepaskhah and Razzaghi, 2009; Dai et al., 2011):

$$PET = 0.0408 \times 0.0023 \times Ra \times (T_{mean} + 17.8) \times (T_{max} - T_{min})^{0.5} \quad (2)$$

where T_{mean} , T_{max} and T_{min} are daily average, daily maximum and daily minimum temperature ($^{\circ}\text{C}$), respectively; and Ra is extraterrestrial radiation ($\text{MJ m}^{-2} \text{d}^{-1}$) for daily period estimated (Allen et al., 1998) as

$$Ra = 24(60)/\pi \times G_{sc} \times dr \times [\omega s \times \sin(\varphi) \times \sin(\delta) + \cos(\varphi) \times \cos(\delta) \times \sin(\omega s)] \quad (3)$$

where G_{sc} is the solar constant, dr is inverse relative distance Earth-Sun, ωs is sunset hour angle, φ is the latitude of the study site and δ is solar declination.

Accordingly, daily mean WT (mWT) was

$$mWT = Ht + \Delta P \div h_p \quad (4)$$

where Ht is daily mean tidal height estimated using equations in Schureman (1941); h_p is number of hours of rainfall during a raining day and $h_p = 1$ if $h_p < 1$, estimated as

$$h_p = P_r \div P_h \quad (5)$$

where P_r is precipitation (cm d^{-1}); P_h is the assumed precipitation rate (cm h^{-1}), P_h is 0.5 cm if daily P_r is less than or equal to 12 cm, otherwise, P_h was P_r divided by 24. We assumed also that mWT was equal to MHL if the calculated mWT was larger than MHL. Similarly, mWT was equal to MLL if mWT was less than MLL.

2.4. Model validation and application

The model validation was divided into two parts: (a) aboveground biomass, based on eight plots (Table 1), and (b) C stocks and fluxes from the mangroves at the ENP. The model performance was evaluated using four widely used quantitative methods (see Model Performance Evaluation below).

The validated model was then applied to simulate C dynamics of the mangrove in the TX, LA and FL using the site specific eco-environmental conditions (Table 2). The model was also run, using those sites as the basis, to assess likely changes under managed conditions, specifically to determine whether forest management alters C sequestration in these forests.

2.5. Model parameterization

The model was parameterized using climate data from Daymet (Thornton et al., 2012). Specific soil and vegetation characteristics from each of the validation plots (Table 1) and application plots (Table 2) were used. Other information needed to initialize the model include mangrove physiological characteristics, such as photosynthetic rate, capacity and temperature, as well as other ecological drivers, such as pore water salinity (Dai et al., 2018). Assumptions for this simulations, except for those used for calculating WT (see Water Table Data), were: (1) land cover at these sites have been mangrove-dominated over the long-term (e.g. > 50 years); (2) there haven't been strong disturbances at the sites within the simulation period at these sites; and (3) that stands were naturally regenerated, and developed without forest management.

This model has functionality to assess effects of disturbances on C sequestration in mangroves. These disturbances may include extreme storms, tsunami, and forest management such as planting, harvesting, drainage and fertilization. To test the behavior of the model for managed forest conditions, we used the plots from TX, LA and FL parameterized as previously described, with the following modification.

Stand stocking reflected full canopy closure, which is characteristic of planted tracts. The actual proportion of canopy cover were 77.2%, 79.4% and 61.8% for FL, LA and TX sites, respectively; hence the managed forest simulation reflected full stocking or 100% canopy cover. We also assumed that available N and P did not constrain tree growth for the managed forest simulation. Other factor were the same as those used for the unmanaged simulation; thereby providing a basis for comparing the differences in mangrove C dynamics at these three sites between unmanaged and managed scenarios using common ecological and environmental conditions.

MCAT-DNDC was run for a 200-year period starting from 1815 at daily time step. At year 1 the mangroves were regenerated, commencing stand development. The simulation period was considered sufficiently long to ensure development of the stands to maturity and providing a common basis for comparing responses across sites.

2.6. Model performance evaluation

Four widely used methods were employed to assess model performance: the coefficient of determination (R^2 , squared correlation coefficient), model performance efficiency (E) (Nash and Sutcliffe, 1970), percent bias (PBIAS), and the RRS [the ratio of the root mean squared error (RMSE) to SD (standard deviation)], to prevent any bias in a single evaluation variable, especially the bias from R^2 and PBIAS.

E (range: from $-\infty$ to 1) is the key variable used to evaluate the model performance, calculated as

$$E = 1 - \frac{\sum (O_i - P_i)^2}{\sum (O_i - \bar{O})^2} \quad (6)$$

where O_i , \bar{O} and P_i are observed values, observation mean and simulated results, respectively. The other evaluation variables, PBIAS and RRS, are computed, respectively, as

$$PBIAS = \frac{\sum (O_i - P_i)}{\sum O_i} \times 100 \quad (7)$$

$$RRS = \frac{RMSE}{SD} \quad (8)$$

where SD is the observation standard deviation; RMSE is the root mean squared error between observation and simulation, the equation is

$$RMSE = \sqrt{\frac{\sum (O_i - P_i)^2}{n}} \quad (9)$$

where n is the number of samples, or pairs of observed and simulated values.

3. Results and discussion

3.1. Water table estimation

The predicted daily WT from ENP followed the observed trends (Fig. 2a) in 2002. The mean of measured daily WT (3.59 cm) was close to the calculated (3.56 cm), and the calculated daily WT was significantly correlated to the observed ($P < 0.01$) with a PBIAS of 0.79% that was within the reasonable error range (-25% , $+25\%$), however, their correspondence was less than satisfactory based on the key evaluation variables E ($-\infty$, 1) and RRS (0, 0.7), supported that E (-0.56) and RRS (1.19) were not within reasonable rating ranges (Moriasi et al., 2007; Dai et al., 2014). Our assumption that the calculated minimum WT could not be lower than the long-term observation of MLL and the predicted maximum WT could not be higher than the long-term observation of MHL could be a factor in the discrepancy in the simulating the water table dynamics. Correspondingly, the calculated minimum and maximum WTs were not consistent with the observed lowest and highest values although the mean WTs were similar. While more accurate simulation of tidally mediated water table dynamics is best done

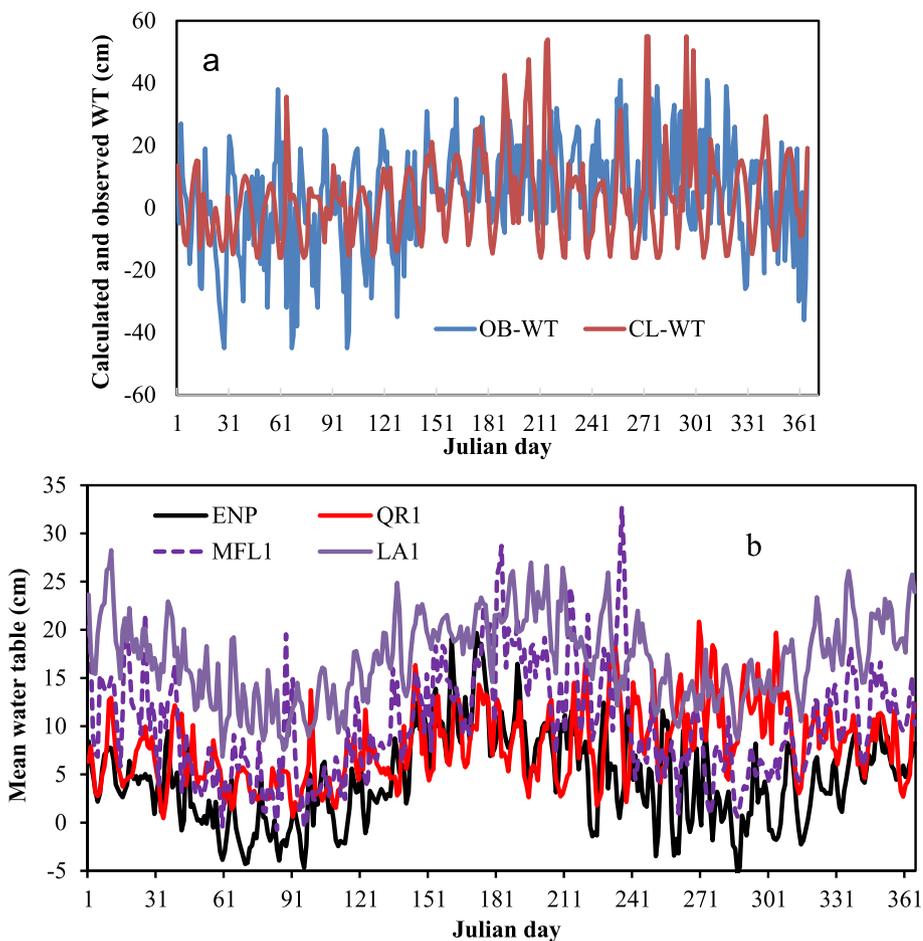


Fig. 2. a. Observed (OB-) vs calculated (CL-) water table depth (WT) in 2002 for Everglades National Park (ENP) site; the observed WT is reported in Castaneda-Moya et al. (2013). WT referenced to the soil surface. b. Calculated mean daily water table depths in 2010 for plots in Everglades National Park (ENP), Quintana Roo (QR1), Louisiana (LA1) and Florida (FL1). The WT is referenced to the soil surface.

using a mechanistic model (Twilley and Rivera-Monroy, 2005; Yang et al., 2010; Haines, 2013), these results provide a functional basis to examine the sensitivity of C dynamics in mangroves to tidally mediated hydrology.

The effect of geographic location on WT was assessed because of differences in climate and tides among the sites. The calculated daily mean WT in 2010 showed substantial differences among sites (Fig. 2b). The difference in calculated annual mean WT within a site for a 10-year period from 2001 to 2010 is relatively small (see Supplement 1), while the relatively large difference among sites remains consistent. The differences in WT among the sites is attributed to the hydro-geomorphic setting that influences tides and ground water and precipitation; the mean tidal levels for a 10-year period from 2001 to 2010 among the sites were 47.5 ± 62.5 , 14.5 ± 21.5 , 22.5 ± 25.5 , 20.0 ± 30.0 and 19.5 ± 35.5 cm for FL1, LA1, TX1, QR1 and ENP, respectively. The WT fluctuation year-to-year within a site (see Supplement 1) is due primarily to climate variability.

To assess the sensitivity of mangrove C pools and fluxes to WT, we simulated AGB, GPP, NPP, LPD (leaf production), DIC (dissolved inorganic carbon), DOC (dissolved organic carbon), POC (particulate organic carbon) and CH_4 for ENP in 2002 using measured and calculated WT; the results were strongly correlated [regression slope of 0.94, PBIAS = 1.12%, RRS = 0.15, E = 0.97 and $R^2 = 0.98$ ($n = 8$, $P < 0.001$)]. Ratios of the simulated C values using predicted versus measured WT values showed that AGB, GPP, NPP and LPD were similar (see Supplement 2), with the results from the simulation using the calculated WT approximately 2.4, 0.8, -0.5 and 2.0% higher than that predicted using the observed WT for those four components, respectively. However, there were large differences in gaseous and aquatic C components (CH_4 , DIC, DOC and POC) between the simulations using

calculated and observed water table depths. The simulated fluxes using the calculated WT were overestimated for DOC (70.6%) and CH_4 (26.5%), and underestimated for DIC (19.0%) and POC (13.2%). These differences reflect that WT is an important factor for assessing gaseous and aquatic C fluxes in mangrove ecosystems. Accordingly, while there may be small differences in biomass estimation between using the calculated and observed WTs, there are large differences in gaseous and aquatic C (CH_4 , DIC, DOC and POC) fluxes, demonstrating that WT observations or data derived from mechanistic hydrological model is necessary to accurately estimate those gaseous and aquatic C fluxes from mangroves.

3.2. Model validation

The results from model validation using data from the Everglades National Park and Quintana Roo, Mexico (Table 3 and Fig. 3) showed

Table 3

Statistical results for model validation using observations from eight sample plots*.

Using aboveground biomass				Using C components for ENP			
R^2	0.99	E	0.99	R^2	0.96	E	0.93
PBIAS	-4.09	RRS	0.09	PBIAS	5.41	RRS	0.22
a	1.06	b	-1.32	a	0.84	b	8.17
MO	89.80	MS	93.48	MO	80.44	MS	75.66

*: R^2 , coefficient of determination (squared correlation coefficient); E, model performance efficiency (Nash and Sutcliffe, 1970); PBIAS, percent bias; RRS, ratio of root mean squared error to standard deviation; a and b , slope and intercept of regression model between observation and simulation, respectively; MO and MS, observed and simulated mean, respectively.

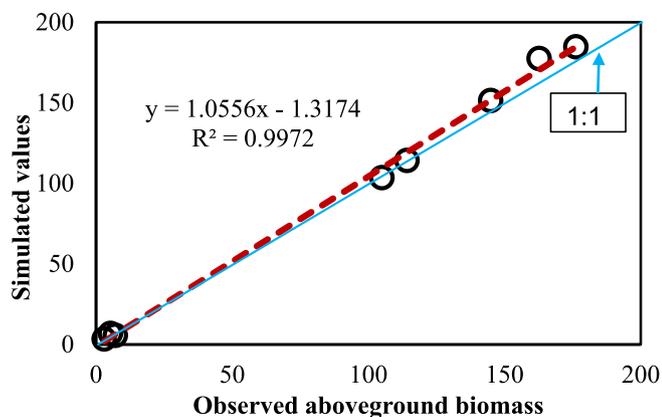


Fig. 3. Observed vs. simulated aboveground biomass (Mg ha^{-1}) for the plots in the Everglades National Park, Florida and Quintana Roo, Mexico; the red dash line is fitted and the light blue line is 1:1. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

that the simulated AGB in good agreement with the observations (Fig. 3; $R^2 = 0.99$). The slope of the regression model between observation and simulation was 1.06, and the intercept was approximately 1.4% of the mean of the measurements. The simulated mean AGB (93.5 Mg ha^{-1}) was close to the observed average (89.8 Mg ha^{-1}). The high performance efficiency of MCAT-DNDC to assess AGB for mangroves at the Everglades National Park and Quintana Roo, Mexican is similar to results reported from simulations using a Mixed Effects Models to estimate C stocks for sites in south Thailand and northern Vietnam (Bukoski et al., 2017).

The comparison of the simulated and observed values for ten C pools and fluxes at ENP (Fig. 4) showed that the simulated values were consistent with the observations with $R^2 = 0.95$, regression slope of 0.84 and intercept of 8.17. However, there were some differences between the observations and simulation results. The simulated results for AGB, DOC, CH_4 , BC (burial C), LLT (leaf litter) and TLT (total litter) were 9.1, 4.4, 10.0, 11.4, 3.2 and 14.1% higher than the observations. For ANPP (aboveground components of NPP, dry matter in $\text{Mg ha}^{-1} \text{ yr}^{-1}$), DIC, POC and Rep (reproductive organ mass, $\text{kg ha}^{-1} \text{ yr}^{-1}$) were 0.8, 22.4, 12.0, 14.4% lower than those observed values.

The simulated BC, CH_4 , DIC, DOC, POC and ANPP for ENP were similar to or within the ranges of observations/estimations reported by Bartlett et al. (1989), Romigh et al. (2006), Barr et al. (2010) and Castaneda-Moya et al. (2013), and the simulated AGB for ENP and QR was in good agreement with the values observed by Castaneda-Moya et al. (2013) and Adame et al. (2013). The model performance

efficiency [$E \in (-\infty, 1)$; Nash and Sutcliffe, 1970] was ≥ 0.98 for the validation against AGB and ≥ 0.93 against ten C components (Table 3), thus, the model performance was within “very good” rating range ($E > 0.75$) (Moriassi et al., 2007; Dai et al., 2014). The PBIAS [$\in(-25, 25)$] was -0.88 for AGB and 8.01 for the ten C components, respectively; and RRS [$\in(0, 0.7)$] was 0.08 for AGB and 0.24 for the ten C components, respectively. These four model evaluation variables conclude consistently that MCAT-DNDC can perform well to assess mangrove C dynamics for sites at ENP and QR.

The simulated DIC for ENP (see Supplement 3) was substantially lower than the observed geometric mean of $308.5 \text{ g C m}^{-2} \text{ yr}^{-1}$, calculated based on the wide observation range ($170\text{--}560 \text{ g C m}^{-2} \text{ yr}^{-1}$) reported by Romigh et al. (2006) and Bouillon et al. (2008) for this plot, although the simulated DIC ($239.5 \text{ g C m}^{-2} \text{ yr}^{-1}$) was within the observed/estimated range. Similarly, simulated POC ($96.0 \text{ g C m}^{-2} \text{ yr}^{-1}$) for ENP was within the observed range ($64\text{--}186 \text{ g C m}^{-2} \text{ yr}^{-1}$), about 12% lower than the geometric mean of the observations. The lower simulated DIC and POC than observed/estimated geometric means for ENP were affected by the wide observed range of hydrological conditions, which are regulated by tides, precipitation and freshwater input. The response is consistent with reports that the variation in hydrology can influence DIC, DOC and POC fluxes (Young et al., 2005; Maher et al., 2013). A limitation in our hydrological data was that there was only a single year of measured water table depths for ENP (2002), hence the uncertainties associated with the other years are unknown, but likely to be high given the correspondence of the single year comparison (Fig. 2a). Accordingly, an accurate representation of the water table position is important to estimating the production and flux of DIC, DOC and POC. There is also feedback to the C burial rate, which is sensitive to the estimates of C export.

Simulated ANPP for ENP (Supplement 3; $14.8 \text{ Mg dry matter ha}^{-1} \text{ yr}^{-1}$) was close to the $14.5 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ reported by Castaneda-Moya et al. (2013), and DOC ($58.4 \text{ g C m}^{-2} \text{ yr}^{-1}$) was similar to the estimated value ($56.0 \text{ g C m}^{-2} \text{ yr}^{-1}$) in Barr et al. (2010). The LLT (dry mass) simulated for ENP was about $11.57 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, 15.4% higher than the mean litter ($10.03 \pm 2.00 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) from mixed, fringe and over-wash, and riverine mangroves, but close to the mean (11.58 ± 1.75) from only riverine mangroves in Florida reported by Twilley et al. (1986), and 8.8% higher than the value ($10.14 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) reported by Castaneda-Moya et al. (2013) for the same plot. Higher simulated LLT for ENP than the observations from other sites in Florida is due mainly to discrepancies in ecological drivers among the sites. ANPP was overestimated by about $300 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for ENP, which might be related to the overestimation of LLT for this site.

Smoak et al. (2013) reported long-term mean C burial rates for a mangrove site, located $\sim 4 \text{ km}$ from the mouth of the Shark River, at about 25.3641° N , 81.0785° W , close to the ENP plot at 25.3646° N ,

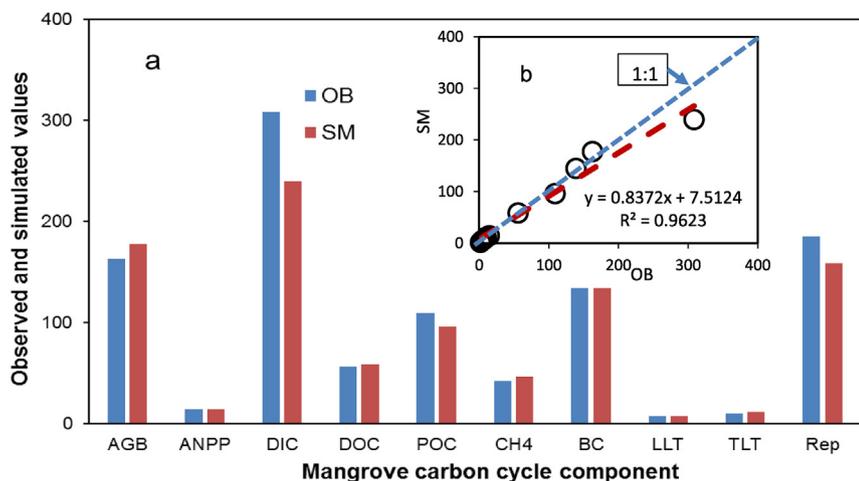


Fig. 4. (a) Comparison of observation and simulation results for ten mangrove carbon cycle components at the Everglades National Park, Florida. AGB (above-ground biomass, Mg ha^{-1}), ANPP (Above-ground net primary productivity; $\text{Mg ha}^{-1} \text{ yr}^{-1}$), CH_4 (CH_4 efflux, $\text{mg ha}^{-1} \text{ d}^{-1}$), LLT (leaf litter, $\text{Mg ha}^{-1} \text{ yr}^{-1}$) and TLT (total litter, $\text{Mg ha}^{-1} \text{ yr}^{-1}$), BC (burial C, $\text{g C m}^{-2} \text{ yr}^{-1}$), DIC (dissolved inorganic C, $\text{g C m}^{-2} \text{ yr}^{-1}$), DOC (dissolved organic C, $\text{g C m}^{-2} \text{ yr}^{-1}$), and POC (particulate organic C, $\text{g C m}^{-2} \text{ yr}^{-1}$), and Rep (reproductive organ mass, $\text{g m}^{-2} \text{ yr}^{-1}$); (b) linear regression of observed versus simulated values for the ten variables at ENP.

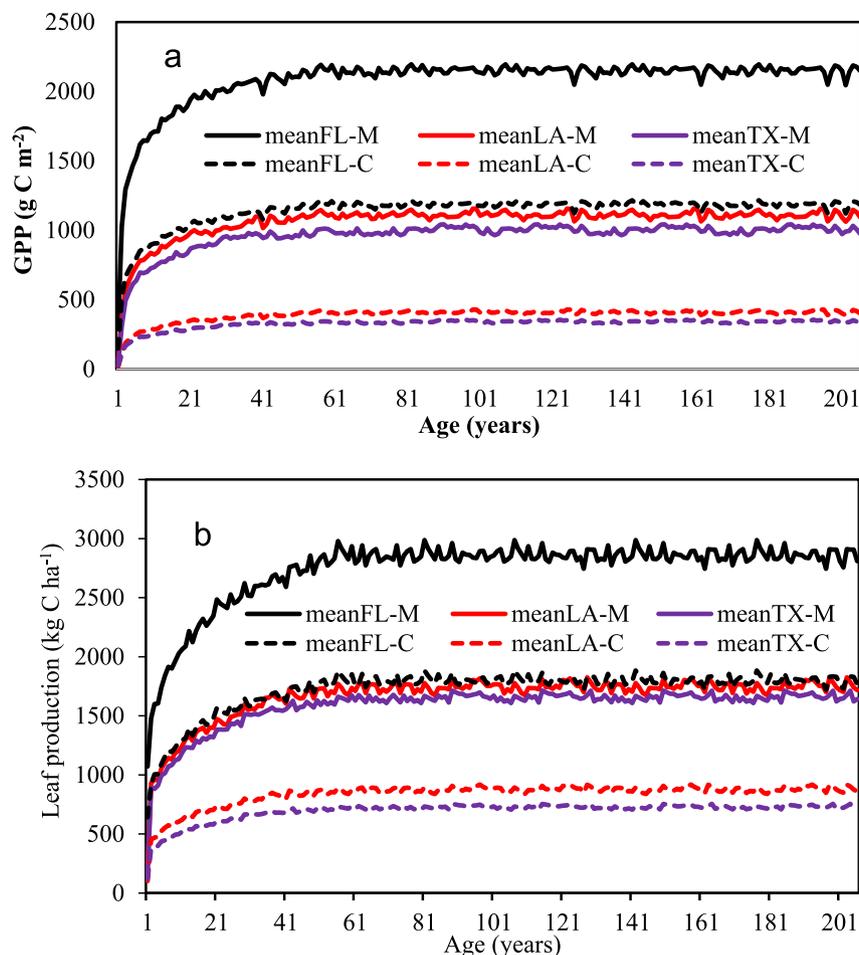


Fig. 5. a. Mean annual gross primary productivity (GPP) of mangroves in Florida (FL), Louisiana (LA) and Texas (TX), simulated for current, unmanaged conditions (-C, dash lines) and managed conditions (-M, solid lines). b. Mean annual leaf production (LPD) under managed (-M, solid lines) and current (-C, dash lines) ecological conditions of mangroves in Florida (FL), Louisiana (LA) and Texas (TX). c. Mean aboveground biomass (AGB) under managed (-M, solid lines) and current (-C, dash lines) ecological conditions of mangroves in Florida (FL), Louisiana (LA) and Texas (TX). d. Mean aboveground net primary productivity (ANPP) under managed (-M, solid lines) and current (-C, dash lines) ecological conditions of mangroves in Florida (FL), Louisiana (LA) and Texas (TX).

81.0779° W (Castaneda-Moya et al., 2013). Their mean C burial rate was $139 \text{ g m}^{-2} \text{ yr}^{-1}$ for a time period from 1924 to 2000, and $151 \text{ g m}^{-2} \text{ yr}^{-1}$ for the time period from 1924 to 2009. The simulated mean C burial rate for ENP was $144.8 \text{ g m}^{-2} \text{ yr}^{-1}$ for the last 120 years from 1895 to 2014 was consistent with the field values found by Smoak et al. (2013). The burial rate from this simulation for ENP was about 4.2% higher than the rate of Smoak et al. (2013) for the time period 1924–2000, but 4.1% lower than their rate for the time period 1924–2009. The lower burial rate simulated for the period 1924–2009 did not consider disturbances. Therefore it did not include Hurricane Katrina that hit the Gulf Coast in 2005. The higher simulated burial rate for the period before 2000 can cause an underestimation of POC for ENP (i.e., the simulated POC for ENP was about 12% lower than the observed mean for the time period 1924–2000), reflecting the partitioning of litter between those two pools.

3.3. Model application

GPP at FL was substantially higher than that at LA and TX (Fig. 5a). All sites approached maximum GPP within a few decades as the effective leaf area for photosynthesis stabilized. Similarly, there was a substantial difference in NPP among the three sites. NPP exhibited the approximate trends and ranking among sites (see Supplement 4). The effect of the managed mangrove scenario is evident when compared to the natural, unmanaged condition. Although the trends in GPP and NPP

with a change in time (e.g., stand age) were similar under the managed and current ecological or unmanaged conditions, C sequestration under the managed ecological condition was substantially higher than that under the current condition. The differences in C sequestration in FL, LA and TX between managed and current ecological conditions demonstrates that model is sensitive to factors affected by forest management (e.g., stocking and site fertility).

The temporal patterns of both GPP and NPP for the sites in FL, LA and TX under different ecological conditions were similar. GPP or NPP significantly increased non-linearly with an increment in stand age, the relationship between the stand age and GPP or NPP was quartic (fourth degree) polynomial ($F > 73,000, P < 0.001$), i.e.,

$$GPP \text{ or } NPP = k_0 \times Age + \sum_1^m k_j \times (\ln(Age))^j \quad (10)$$

where *Age* is the stand age; k_0 and k_j are coefficients, $j = 1, 2, \dots, m$, $m = 4$. However, these coefficients are different among the sites and between GPP and NPP.

Annual leaf production (LPD) for each of the sites in FL, LA and TX showed that the differences in LPD among the sites (Fig. 5b) and the changing trend in the annual production at each site were following the pattern of GPP, i.e., the LPD at FL site was substantially higher than the values at LA and TX sites, and after the annual production reached its peak, it kept relatively stable with inter-annual fluctuations caused by climate undulation year-to-year.

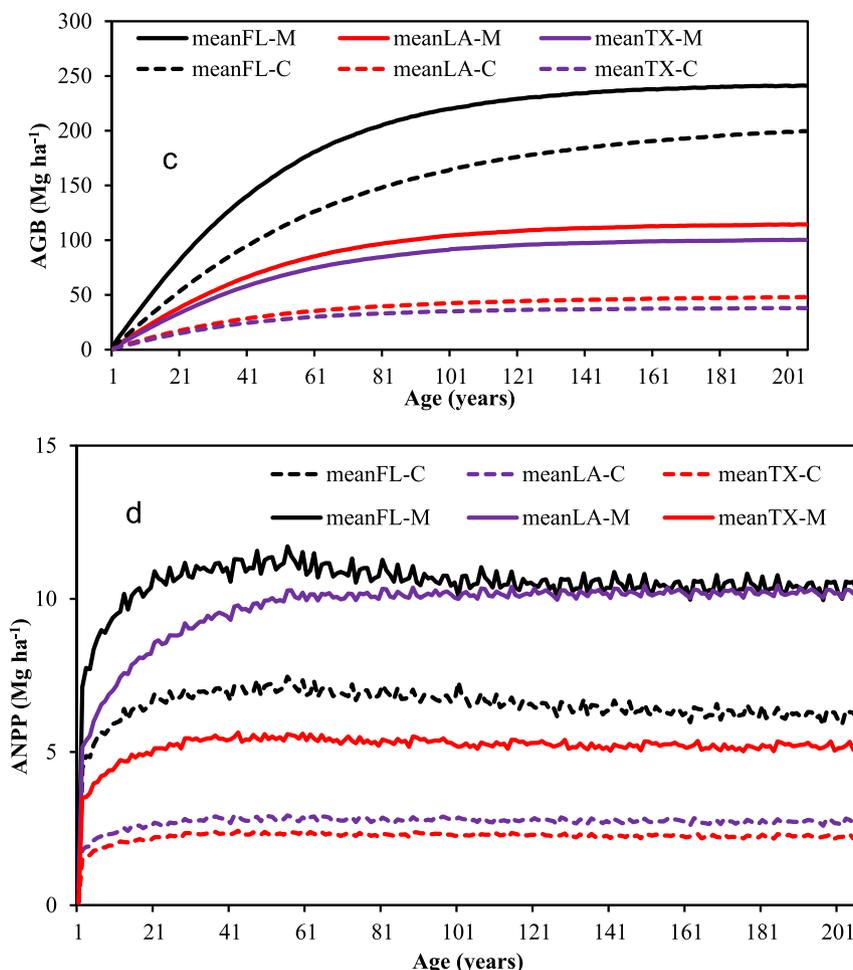


Fig. 5. (continued)

Soil salinity can be one of important factors affecting C sequestration in mangroves. For example, the long-term mean temperature at FL (21.1 C) and LA (20.9 C) was similar, and the current mean tree cover rate at these two sites was approximate (77.2% at FL; 79.4% at LA), however, the AGB at FL was higher than that at LA (Fig. 5c). This is because the soil salinity at LA (44.4 ppt) was higher than the salinity at FL (34.7 ppt).

There were small differences in the relationship between AGB and the stand age among the sites in FL, LA and TX (Fig. 5c). Unlike the relationship between stand age and GPP or NPP, AGB was not logarithmically correlated to the stand age, i.e., the correlation can be described as follows

$$AGB_{Age} = \sum_1^m k_i \times (Age)^i \quad (11)$$

where k_i is site specific coefficient; $i = 1, 2, \dots, m$. m is equal to 4 for FL site under managed ecological condition, and to 3 for all other sites under both managed and current mangrove situations. Similarly, Eq. (11) can be used to describe the relationship between AGB and stand age for ENP and all plots in QR with plot specific coefficients, and m is equal to 4 for all eight plots.

There were differences in simulated ANPP for the sites in FL, LA and TX under current and managed ecological conditions (Fig. 5d); however, the difference in ANPP between FL and LA under managed condition was small. Although AGB at LA was slightly higher than at TX, ANPP at LA was substantially higher than that at TX under current and managed conditions due mainly to higher litterfall at LA than at TX caused by freezing (average of 4.95 days yr⁻¹ at LA and 2.66 days yr⁻¹

at TX) based on Daymet climate data in the 35 years period from 1980 to 2014.

The relationships between stand age and production (GPP, NPP) or allocation (AGB, LPD) among the three sites were similar, despite the substantial differences in site and environmental conditions, and vegetation stature that ranged from dwarf to tall mangroves. However, the C sequestration rate was substantially different among sites due to differences in ecological conditions, including soil salinity, vegetation, soil phosphorous and climate. The relationship between NPP and stand age from this study indicates that long term C sequestration in mangroves may be different from that in the upland forests. NPP in mangroves decreases only slightly, if at all, with an increase in stand age after it reaches its peak, but NPP in some upland forests can decrease substantially with an increase in stand age after its peak (He et al., 2012; Dai et al., 2014). This difference in C sequestration may reflect a sustained capacity in mangroves to sequester C throughout the duration of the stand.

The relationships between AGB and stand age simulated for nine plots at FL and the nine plots at TX (Fig. 6a and Fig. 6b) exhibited that there were some divergences in C sequestration not only among plots at each site, but also among sites. This difference in AGB among the plots within a site is mainly related to soil salinity (Ball and Pidsley, 1995), caused by micro-geography that can lead distinct impacts of tides and freshwater inputs on the hydrology among the mangrove plots, indicating that topographic gradient and micro-geographical settings can influence C sequestration in mangroves (Alongi, 2009). For example, there were some small differences in AGB among plots in FL because the differences in salinity were small (ranged from 28.83 to 52.30 ppt with a mean of 34.73 ppt); however, the differences in salinity among the

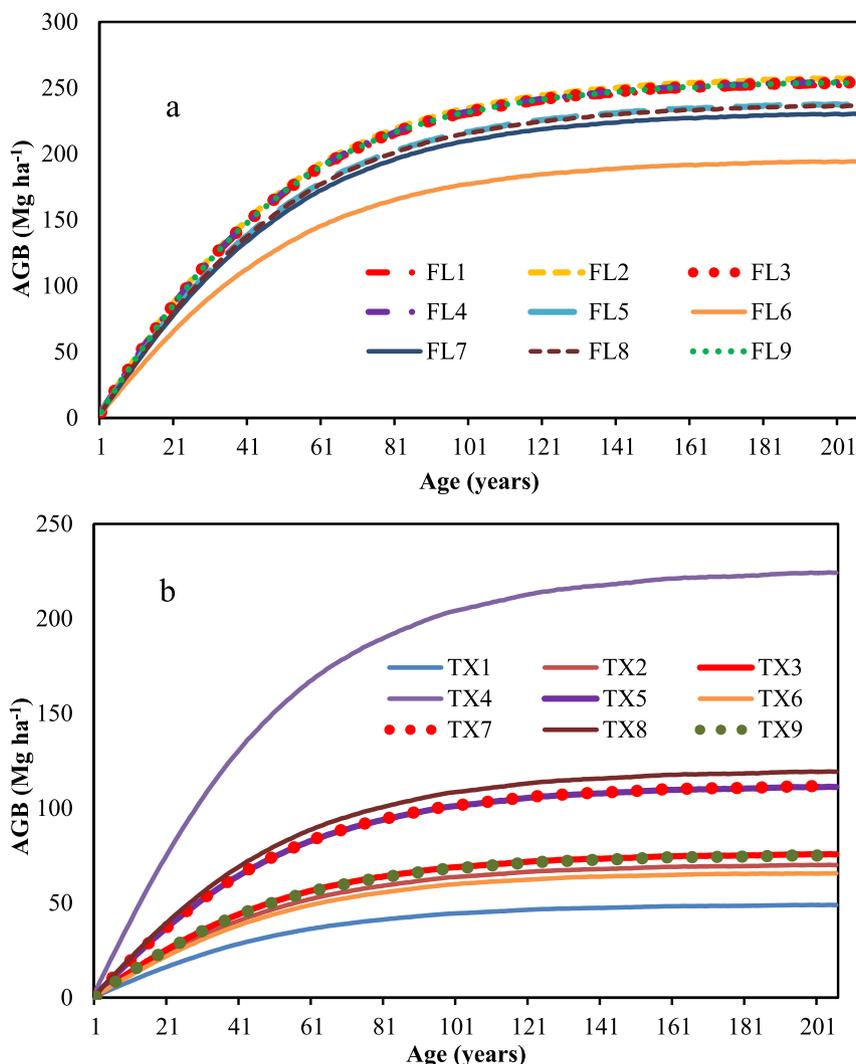


Fig. 6. Difference in AGB (aboveground biomass) at different plots in Florida (a) and Texas (b).

plots in TX were large (between 37.53 and 79.7 ppt with a mean of 52.46 ppt) such that the differences in AGB among the plots in TX were large too. Similarly, the simulated GPP and NPP were substantially higher at FL than those at LA and TX, indicating that soil salinity can be a significant influence C sequestration in mangroves.

The simulated means of AGB for FL, LA and TX under managed ecological condition with observed salinity (34.73 ± 7.49 ppt, 44.36 ± 7.59 ppt and 52.46 ± 12.72 ppt for FL, LA and TX, respectively) were 95.6, 45.2 and 39.6 Mg C ha^{-1} , respectively, within the simulation period, indicating that AGB decreased with an increase in salinity. The relationship between salinity and AGB in coastal south USA appears be non-linear ($R^2 = 0.5073$, $n = 27$, $P < 0.001$), following

$$AGB = k \times S^{-1.537} \quad (12)$$

where k is the coefficient and S is salinity in ppt.

The difference in C sequestration in mangroves between the simulated managed forest and current unmanaged conditions for the three sites in FL, LA and TX indicates that management can increase C sequestration on those sites by improving stocking and ameliorating possible N and P deficiencies at the LA and TX sites. The AGB under the managed forest scenario was approximately 26, 58 and 60% higher than under the current condition. The larger increase in AGB at LA and TX than at FL suggests that soil N and P management can substantially increase C sequestration because the differences in the cover rate

among the three sites were small. However, this forest management scenario did not change effects of climate, soil salinity and hydrology on C dynamics.

4. Conclusion

The model validation demonstrated that MCAT-DNDC can be used to estimate mangrove biomass dynamics in the southeastern USA and Mexican Caribbean with high model performance efficiency. The comparison of ten simulated C pools and fluxes with measured data from mangroves in the ENP affirmed that the model is particularly effective in representing AGB as well as C fluxes into the air and soil water. The model effectively reflects the influence of ecological drivers on C dynamics in mangroves; those of particular importance include soil salinity, hydrological conditions, and vegetation cover (e.g., leaf area). The model appears to be particularly sensitive to the water table dynamics, as evidenced by significant differences in CH_4 , DIC, DOC and POC when predicted using estimated WT as opposed to measured WT. Accordingly, water table data to support the simulations would preferably be obtained from long-term hydrological data bases or a process-based hydrological model. However, simulated AGB, GPP, NPP and LPD did not appear to be as sensitive to the differences in measured water table and the simplified approach used to estimate water table for this study. The simulations of AGB reflected differences in ecological conditions among the sites in Florida, Louisiana, Texas, and Mexico,

although the functional relationship of AGB to stand age was similar. Correspondingly, GPP, NPP and LPD were similarly related to stand age.

The simulations also demonstrated that the model may be useful in considering how management could affect C sequestration. The management scenario illustrated the varied effect on C sequestration among the sites in FL, LA and TX, reflecting the interactions of site conditions with management specifications. Accordingly, the stability of the model and the reasonable predictions to a management scenario suggest that MCAT-DNDC may be a useful tool for considering how management or restoration of mangroves could sustain or enhance C sequestration and long-term storage in the soil. Further analyses using data from actively managed sites are warranted to affirm the applications.

The mangrove C dynamics simulated by MCAT-DNDC appear to be stable across the ranges of sites in the Caribbean basin. However, additional work to validate predictions of pools and fluxes is warranted. The model is an effective tool to assess the sensitivity of the C dynamics to site conditions, hence, it should also be effective for considering the C sequestration potentials for restored or agrgrading sites. This model could be used as a tool in MRV for REDD + to assess long-term C dynamics in mangroves. It also provide capabilities for considering the interactions of changing environmental conditions and extreme events with C dynamics in mangroves. However, the quality of the hydrological data, especially tidal dynamics has a strong influence on the modeled processes; therefore data from high quality measurements or mechanistic hydrology models are warranted.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.ecss.2018.04.036>.

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