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Landcover change in mangroves of Fiji: Implications for climate change mitigation and adaptation in the Pacific

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A R T I C L E I N F O

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A B S T R A C T

Mangrove coverage in Fiji is among the highest of all Pacific island nations. These ecosystems store disproportionate amounts of carbon, provide critically important resources for communities, and protect coastal communities against the impacts of tropical cyclones. They are therefore vital in mitigating and adapting to the impacts of climate change. An improved understanding of both the scale and drivers of mangrove loss in Fiji can underpin sustainable management strategies and achieve climate change mitigation and adaptation goals. In this study we assessed mangrove cover, landcover change, and drivers of landcover change for Fiji between 2001 and 2018, as well as the impacts of landcover change on the structural characteristics of mangroves at selected sites on the Fijian island of Viti Levu. Results were then framed within the context of developing management responses, including the potential to develop forest carbon projects. We found Fiji's mangrove estate to be 65,243 ha, with a loss of 1135 ha between 2001 and 2018 and an annual rate of loss of 0.11%. Tropical cyclones accounted for 77% of loss (~870 ha), with highest losses along the northern coastlines of Viti Levu and Vanua Levu. Mangrove structural characteristics showed high variability in the level of damage incurred, with taller riverine and hinterland vegetation sustaining greater levels of damage than coastal fringing or scrub mangroves. There was no tropical cyclone damage evident along the southern coastline of Viti Levu, with small-scale harvesting the predominate driver of loss in this region. Because of the large effect of cyclone damage on mangroves in the region, small to medium scale restoration projects may be appropriate interventions to increase mangrove cover and carbon stocks. Where harvesting of mangroves occurs, improved management to avoid deforestation could also provide opportunities to maintain mangrove cover and carbon stocks.

1. Introduction

Mangroves provide a vast range of critically important ecosystems services (Barbier et al. 2011; Lau 2013) which support the livelihoods and wellbeing of hundreds of millions of coastal people across the tropics (Mohammed 2012). Of these ecosystem services, perhaps the most discussed in international discourse over the last decade has been the significant role mangroves can play in climate change mitigation and adaptation (Duarte et al., 2013; Cameron et al. 2018). They are among the most productive ecosystems on Earth and exhibit disproportionately high rates of carbon sequestration and storage in comparison to most other ecosystems (Alongi, 2014, Donato et al. 2011). Mangroves have also proven to be critical in mitigating coastlines from the impacts of intense storms and tsunami (Hochard et al., 2019; Vermaat and Thampanya, 2006; Barbier, 2006). However, these ecosystems have been severely degraded with global loss estimated at 30–50% of original extant (Alongi, 2014), although contemporary rates of loss have declined to less than 0.4% per year (Friess et al., 2019). The high productivity and carbon density of many mangrove ecosystems provide multiple incentives to address ongoing loss through the conservation of intact forests and rehabilitation of degraded areas (McLeod et al. 2011). Mangroves and other 'blue carbon' ecosystems such as seagrasses and tidal marshes have been included in climate change mitigation and adaptation commitments to the International Paris Agreement (Herr and Landis, 2017; Adame et al., 2018; Taillardat et al., 2018), within forest carbon offset markets

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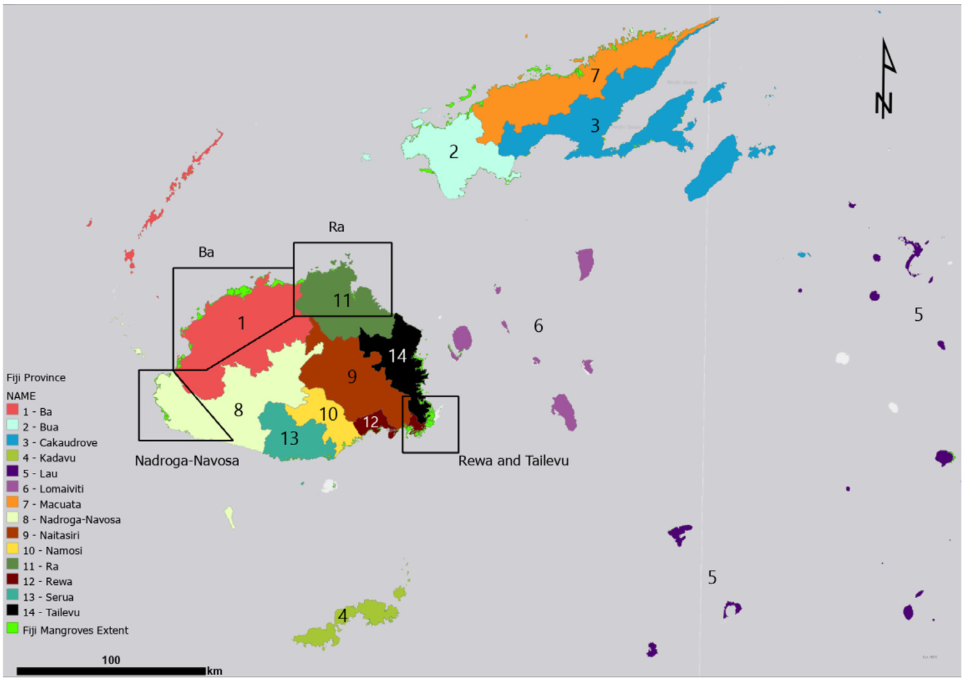

Fig. 1. Mangrove extent in the Fijian archipelago and location of selected provinces assessed to determine drivers of mangrove loss.

(Needleman et al. 2019), as well as other Payment for Ecosystem Services (PES) schemes (Wylie et al., 2013, Locatelli et al., 2014) which provides opportunities for incentivising conservation and restoration.

Recognising the importance of mangroves for climate change mitigation and adaption as well as the potential of blue carbon projects to support the management of mangroves, the Fijian Government has identified the need to reverse ongoing mangrove losses, conserve and sustainably manage mangroves, and account for the ecosystem service values of mangroves in national climate strategies and mechanisms (MoE 2018). For example, Fiji currently has a Mangrove Management Plan developed under the 2013 Mangrove Ecosystems for Climate Change Adaptation and Livelihoods Project, although this plan is yet to be formally implemented. While mangroves are included within the Environmental Management Plan (2005) and Protected Species Act (2002) of Fiji and have recently been included as priority habitats for conservation and restoration under Fiji's Low Emission Development Strategy 2018-2050 (MoE 2018), the country's Nationally Determined Contribution Implementation Roadmap 2017-2030 (MoE 2017) is focused on the energy sector (i.e. CO_2 emissions from electricity, industry and transport). However, the important role of mangroves in climate change mitigation and adaption is increasingly recognised in policy discourse among island nations of the wider Pacific, particularly with regards to potential buffering against imminent threats such as sea-level rise (Ellison, 2018).

To support the Fijian Governments emphasis on blue carbon and to incorporate mangroves more broadly within the forest carbon inventory in Fiji requires an improved understanding of both the scale and drivers of mangrove loss. Mangrove coverage and estimates of loss for Fiji have been included in a range of coarse scale global assessments, including the Mangrove Forests of the World (MFW) database (Giri et al., 2011) which estimated mangrove spatial extant at ~40, 000 ha. Hamilton and Casey (2016) estimated Fiji has lost 0.19% of mangroves (77 ha) between 2000 and 2012 at an annual loss rate of 0.02% using the MFW database, which was among the lowest rates of loss globally. More recently, Worthington and Spalding (2019) estimated mangrove coverage in Fiji at 50, 968 ha with an area of loss of 637 ha since 1996. However, there are no national scale remote sensing assessments of mangrove cover and landcover change, verified by field studies, specific to Fiji. We addressed this critical knowledge gap in the current study by assessing (a) the current extent of mangrove coverage, landcover change, and drivers of landcover change in Fiji between 2001 and 2018; and (b) mangrove forest structure and the impacts of landcover change on structural characteristics of mangroves at selected sites. Results from this analysis are then framed within the context of assessing the feasibility of developing blue carbon projects in Fiji. The research presented here helps improve the knowledge base from which to inform decision making for the management and conservation of mangroves in Fiji and elsewhere in the Pacific region.

2. Materials and methods

2.1. Analysis of mangrove extent, landcover change and drivers of loss

To determine the spatial extent of mangroves and change in extent across the Fijian archipelago, an on-line geographical information system (GIS) datasets from the Global Forest Change (GFC) 2000–2018 (see methods described in Hansen et al., 2013) and Fiji Forest Change Detection 2006–2018 (GIZ, SPC, SPREP, 2019) studies was used. Annual mangrove extent and change from 2000 through 2018 were assessed across all 14 Fijian provinces using a time-series analysis of Landsat composite images from the GFC 10–20° S, 170–180° E granule. Forest loss during this period was defined as a stand-replacement disturbance, or a change from a forest to non-forest state (Hansen et al., 2013). Specific drivers of annual mangrove loss for the selected provinces of Ba, Nadroga-Navosa, Rewa and Tailevu, and Ra were then identified through time series analysis of satellite imagery using platforms such as Google Earth Pro (2019) and Google Earth Engine (2019), with results verified through field surveys at selected sites in 2019 (Fig. 1).

2.2. Field surveys

Field surveys were conducted to support and verify GIS analysis (extent of loss and drivers of landcover change) in four contrasting regions around Viti Levu between March and July 2019. Surveys were conducted in 1) Ba Province which is centered on the north-western corner of Viti Levu; 2) Nadroga-Navosa Province which is located along the western coastline of Viti Levu bound by Nadi township to the north and Sigatoka to the south-east; 3) The Rewa and Tailevu provinces of south

Table 1
Summary of mangrove extent, mangrove coverage loss and drivers of loss for selected provinces in Fiji 2001-2018. The unsurveyed Bua Province of Vanua Levu accounts for an additional ~223.7 ha of loss due to tropical cyclones. N/A = not applicable.

Province	Mangrove extent 2018 (ha)	Mangrove loss 2001–2018 (ha)	% loss (per annum)	Drivers of mangrove loss and estimated extent (ha) and proportion (%)			
				TCs	Tourism development	Dredge disposal	Other
Ba	13,066	343.5	2.6 (0.16)	~210 (61%)	~120 (35%)	~13.5 (4%)	N/A
Nadroga-Navosa	2599	16.2	0.6 (0.04)	~16.2 (100%)	N/A	N/A	N/A
Rewa and Tailevu	11,005.6	105	0.9 (0.1)	~61.1 (58%)	N/A	~18.9 (18%)	~25.1 (24%)
Ra	2271.8	315.2	12.2 (0.76)	~307.7 (98%)	N/A	N/A	~7.5 (2%)
Fiji total	**65,243**	**1,135**	**1.7 (0.11)**	**~870 (77%)**	**~150 (13%)**	**~32.4 (3%)**	**~82.6 (7%)**

– eastern Viti Levu which encompass the Rewa Delta, Fiji's largest mangrove ecosystem; and 4) Ra province along the north-eastern coastline of Viti Levu, Viti Levu Bay, where substantial loss of mangrove cover had been observed. Supplementary Materials Table S1 describes each site in terms of its geomorphic classification and mangrove composition.

Three study regions (Ba Province, Nadroga-Navosa Province, and Rewa and Tailevu provinces which divide the Rewa Delta) were then selected for assessments of mangrove structural characteristics. 12 distinct sites within these regions were assessed *in-situ* with field measurements capturing mangrove forest composition following Kauffman and Donato (2012). Briefly, transects traversing from the landward to seaward edge of the mangrove are tailored to suit site specific conditions and to capture different assemblages across the intertidal zone. Within each transect, up to 6 plots of 7 m radius (154 m^2) were established at ~50 m intervals in which species and diameter at breast height (DBH, measured at 1.3 m above the ground) for all trees within the plots were measured. For scrub mangroves (<2 m tall), diameter was recorded at 30 centimetres above the ground with tree height also recorded. For *Rhizophora* species, which typically exhibit numerous prop roots and may branch off the main stem above 1.3 m in height, DBH was measured above the highest prop root. Nested sub-plots with a 2 m radius were established at the centre of each plot to record seedling and sapling density. The mass of dead and downed woody debris (DDWD) was calculated through the planar intercept technique adapted for mangroves (Kauffman and Donato, 2012). Emanating from the centre of each plot, four 14 m transects were established and offset from each other by 90°. Along the length of each transect, any downed, dead woody material (fallen and detached twigs, branches, prop roots, or tree stems) intercepting the transect was recorded with fragments classified into one of 5 categories; fine (<0.6 cm), small (0.6–2.5 cm), medium (>2.5–7.5 cm), large solid (>7.5 cm) or large rotten (> 7.5 cm). Allometric equations were used to calculate above and below biomass using both species specific equations (where available) and common (mangrove generic) equations (Komiyama et al., 2005). The carbon content of biomass was calculated by multiplying by a factor of 0.464 for aboveground biomass and 0.39 for belowground biomass (Kauffman and Donato, 2012). This methodology enabled data to be captured on key forest structural and biomass attributes such as the prevalence of tree harvesting and volume of dead and downed woody debris resultant from cyclone damage which enabled verification of the drivers of mangrove loss as well as their impacts on mangrove ecosystems.

3. Results

3.1. Mangrove extent, landcover change, and drivers of loss across the Fiji Islands at the provincial level

The areal extent of mangroves in the Fijian archipelago was 65,243 ha, with the majority of coverage evenly dispersed around the coastlines of the two largest islands of Viti Levu (31,509 ha) and Vanua Levu (29,938 ha). At the site level, the largest contiguous areas of mangroves are located in the Rewa (7110 ha) and Ba Delta's (5540 ha) of Viti Levu followed by the Labasa Delta (1545 ha) on Vanua Levu. Some of the coral atoll islands directly offshore from Vanua Levu such as Talailau (690 ha) and Nadogo (1210 ha) are also significant mangrove habitats and are almost exclusively covered in mangroves.

Mangrove loss over the period 2001–2018 across Fiji was estimated at 1,135 hectares, a decrease of 1.7% in cover since 2001 with an average annual rate of loss of 0.11% (Table 1). Provinces exhibiting the highest losses are Ra (315.3 ha, 12.2%), Ba (343.5 ha, 2.6%) and Bua (Vanua Levu, 223.7 ha, 2.3%). In contrast, regions with significant mangrove cover but minimal loss included Cakaudrove and Macuata on the south-east and north-east coasts of Vanua Levu (0.7% and 0.3% loss of cover between 2001-2018 respectively) and Rewa, Nadroga-Navosa and Serua on the south-east, western and southern coast of Viti Levu (0.8%, 0.6% and 0.3% coverage loss between 2001 and 2018 respectively).

Interpretation of annual mangrove cover loss with corresponding satellite imagery data reveals that approximately 77% of loss (~870 ha) can be directly attributed to the successive impacts of Tropical Cyclones (TCs) Gene (Category 3, January 2008), Mick (Category 2, December 2009), Evan (Category 4, December 2012), and Winston (Category 5, February 2016. Fig. 2), with mangrove loss largely concentrated in the Ra, Ba and Bua provinces (Table 1, Fig. 3). These cyclones had maximum sustained wind speeds of 155 km/h, 110 km/h, 185 km/h and 280 km/h respectively (Asia-Pacific Data-Research Center, 2020).

After TCs, the next most significant drivers of coverage loss were the conversion of mangroves for tourism development and coastal reclamation (~120 ha) followed by the disposal of dredging spoil in the Ba and Rewa Deltas (~33 ha). The remaining 112 ha of loss was attributable to smaller scale conversion for industrial estates, squatter housing, agriculture and construction of sugarcane tram lines, as well as harvesting for both fuelwood and construction materials, all of which were previously recognized drivers of mangrove loss in Fiji (MoE 2018).

The Supplementary Materials provide a detailed description of changes in mangrove extent, drivers of loss, and resultant impacts on forest structure for sites assessed during the field survey, including effects from the succession of TCs on mangrove ecosystems of Ba, Nadroga-Navosa and Ra.

3.2. Impacts of tropical cyclones and human activities on mangrove structural attributes at the site level

Mangrove forest structure and the impacts of damage by TCs and human activities varied across the sites assessed in field surveys. At a site-specific scale, the Ba and Ra provinces had losses of mangrove coverage driven principally through the impacts of successive tropical cyclones and together account for almost 60% of loss in Fiji between 2001 and 2018, with mangrove loss negligible in Nadroga-Navosa and Rewa and Tailevu. Structural damage was mostly evident in taller mangroves of Ba $1_{\text{TC damage}}$, Ba $4_{\text{Island TC damage}}$, Tuva $6_{\text{TC damage}}$, and Ra $13_{\text{Hinterland}}$. In contrast, despite the shorter scrub mangroves of Ba 2_{scub}, Ba $3_{\text{Island intact}}$, and Ra $14_{\text{Coastal margin}}$ being directly adjacent to TC damaged sites, there was very little evidence of structural damage. A characteristic of TC affected sites was, as expected, a high proportion of biomass within the downed wood and standing dead tree biomass pools, although the scrub mangroves of Ba 2_{scrub} also had almost half of its biomass vested fine

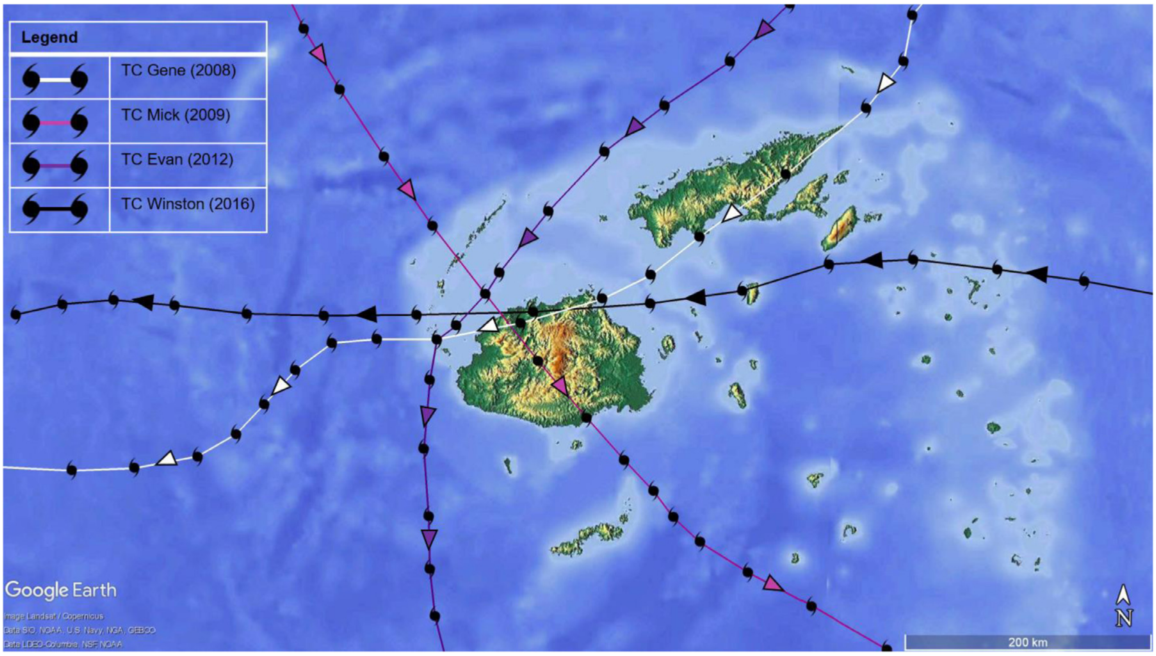

Fig. 2. Trajectories of recent tropical cyclones (TC) to have affected the Fijian Islands. Data sourced from National Oceanic and Atmospheric Administration 2020.

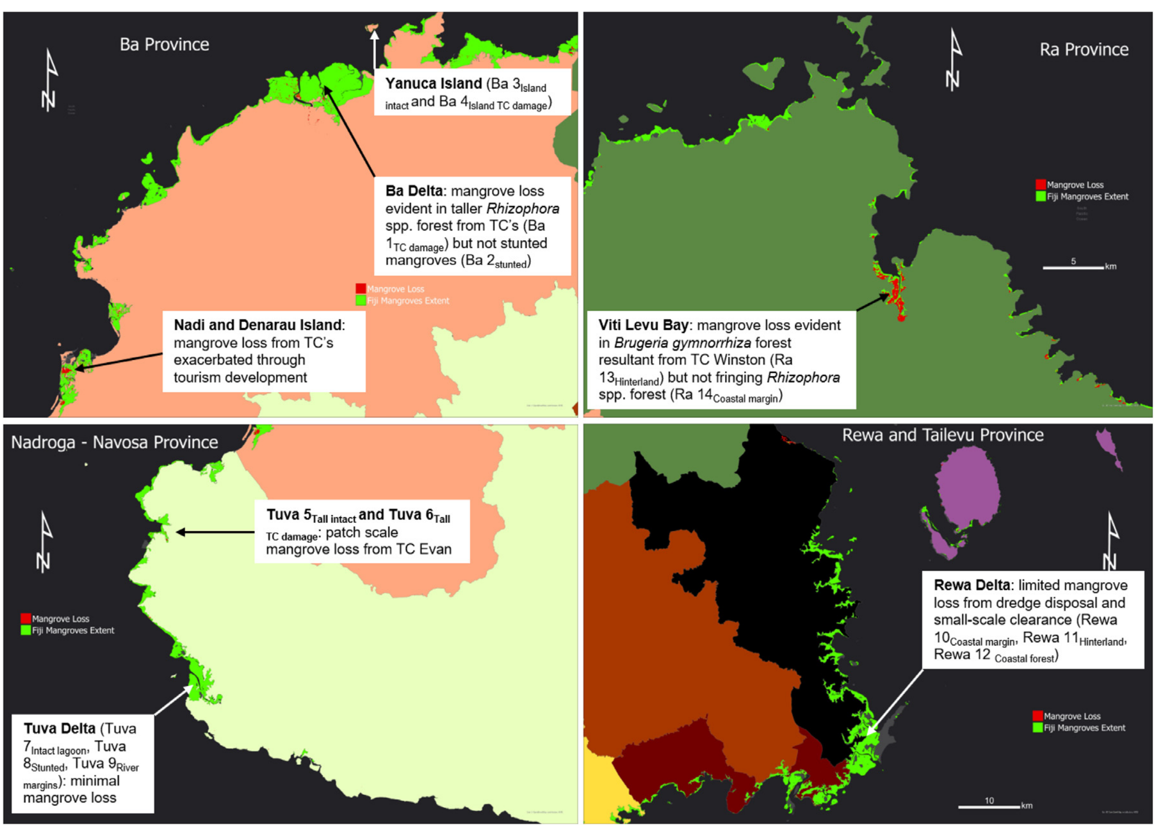

Fig. 3. Mangrove cover loss at Ba, Ra, Rewa and Tailevu, and Nadroga-Navosa provinces on the main island of Viti Levu. Insets explain the main causes of mangrove loss.

woody debris (Table 2). In the Ba Delta, a temporal sequence of imagery (Fig. 4) shows the impacts of successive TCs followed by signs of subsequent recovery.

In sites assessed in Rewa and Tailevu, impacts on mangrove forest structure were caused through human activities related to both selective harvesting and small-scale (i.e. <1 ha) clearance in small dispersed patches, with mangroves extracted for use as fuelwood and timber (Fig. 5, top left). Rewa $11_{\text{Hinterland}}$ and Rewa $12_{\text{Coastal forest}}$ had harvested tree densities of 83.5 ± 24.9 and 59.3 ± 35.3 trees ha^{-1} respectively (Table 2).

Table 2
Structural attributes and carbon stocks of sites and locations assessed during field studies. Values are mean ± standard error and sourced from Cameron et al. (2020). N/A = not applicable. N/M = not measured. DDWD = dead and downed woody debris. Species abbreviations: *R. sty = Rhizophora stylosa; R. sam = Rhizophora samoensis; R. sel = Rhizophora selala; B. gym = Bruguiera gymnorrhiza; E. aga = Excoecaria agallocha.*

Site ID (no. plots)	Key structural attributes							Biomass carbon stocks			
	Species dominance (%)	Canopy height (meters)	Live tree density (trees ha^{-1})	Dead tree density (trees ha^{-1})	Harvested tree density (no. trees harvested ha^{-1})	DBH alive trees (cm)	DBH dead trees (cm)	Live tree biomass (Mg C ha^{-1})	Dead tree biomass (Mg C ha^{-1})	DDWD (Mg C ha^{-1})	Total biomass (Mg C ha^{-1})
Ba											
Ba $1_{\text{TC damage}}$ (9)	*R. sty* (83.7%); *R. sam* (15.1%)	15–20	1724.4 ± 362.7	497.8 ± 78	N/A	6.4 ± 0.8	14.2 ± 1.5	27.5 ± 10.2	29.7 ± 6	145.3 ± 10.7	202.4 ± 27
Ba 2_{Scrub} (6)	*R. sel* (100%)	2–4	4902.6 ± 221.6	N / A	N/A	3.0 ± 0.0	0	4.5 ± 0.3	0	3.7 ± 0.4	8.2 ± 0.7
Ba $3_{\text{Island intact}}$ (4)	*R. sty* (99%); *B. gym* (1%)	5–10	6363.6 ± 690.3	243.5 ± 66.9	N/A	4.0 ± 0.4	9.5 ± 2.2	32.0 ± 8	7.2 ± 4.8	9.7 ± 1.7	48.9 ± 14.4
Ba $4_{\text{Island TC damage}}$ (3)	*R. sty* (100%)	15–20	N / A	1147.2 ± 312.2	N/A	N/A	10.6 ± 0.7	0	23.8 ± 2.9	173.1 ± 10.7	196.8 ± 26.9
Nadroga-Navosa											
Tuva $5_{\text{Tall intact}}$ (3)	*R. sel* (62.2%); *R. sty* (34.4%)	15–20	1948.1 ± 292.8	129.9 ± 64.9	N/A	12.5 ± 0.3	6 ± 1.0	183.1 ± 9.8	0.6 ± 0.3	89.6 ± 18.3	273.3 ± 28.3
Tuva $6_{\text{Tall TC damage}}$ (3)	*R. sty* (65.6%); *R. sel* (34.4%)	15–20	1385.3 ± 21.6	887.4 ± 141.9	N/A	7.8 ± 2.6 [k]	7.4 ± 0.7	59.0 ± 30.6	10.3 ± 2.4	221.3 ± 23.8	290.7 ± 56.8
Tuva $7_{\text{Intact lagoon}}$ (6)	*B. gym* (76.2%); *R. sty* (18%)	15–25	1861.5 ± 188.7 [h]	357.1 ± 131.8	N/A	16.1 ± 2.7 [h]	6.7 ± 3.0	197.2 ± 47	12.2 ± 9.7	70.3 ± 16.4	279.7 ± 73.1
Tuva 8_{Scrub} (7)	*R. sam* (48.2%); *R. sty* (30.1%); *R. sel* (19.3%)	4–6	6902.7 ± 722.1	N/A	N/A	5.2 ± 0.6	0	26.2 ± 12.9	0	N/M	26.2 ± 12.9
Tuva $9_{\text{River margins}}$ (10)	*R. sel* (47.7%); *B. gym* (29.3%); *R. sty* (16.3%)	10–15	1837.7 ± 200.7	N/A	N/A	12.2 ± 1.1	0	100.7 ± 17.5	0	N/M	100.7 ± 17.5
Rewa and Tailevu											
Rewa $10_{\text{Coastal margin}}$ (7)	*R. sty* (64.1%); *B. gym* (34.2%)	10–15	3460.1 ± 1115.7	278.3 ± 146.2	N/A	12.6 ± 2.1	14.2 ± 5.6	154.2 ± 17.5	8.9 ± 5.3	32.5 ± 3.4	195.7 ± 25.9
Rewa $11_{\text{Hinterland}}$ (14)	*B. gym* (76.5%); *E. aga* (9.4%); *R. sty* (7.5%)	15–25	987.9 ± 124.4	N/A	83.5 ± 24.9	25.7 ± 2.4	0	231.7 ± 28.7	0	67.6 ± 7.8	299.3 ± 36.6
Rewa $12_{\text{Coastal forest}}$ (12)	*B. gym* (90.4%); *R. sty* (6.9%)	15–25	1975.1 ± 345.6	N/A	59.5 ± 35.3	17.7 ± 2.4	0	180.1 ± 33.5	0	57.5 ± 17.9	237.6 ± 51.4

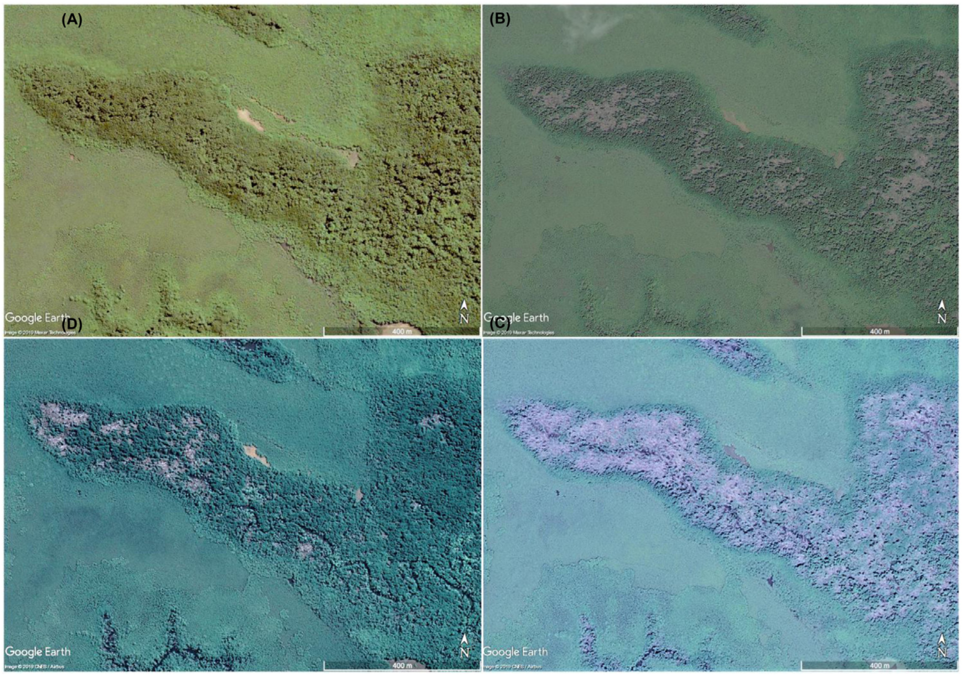

Fig. 4. Clockwise from top left: (A) May 2004 - Healthy, intact mangrove forest within the Ba Delta showing both tall riverine and scrub mangrove communities; (B) January 2011 – Damage to taller riverine mangroves from TCs Gene (Jan 2008) and Mick (Dec 2009); (C) February 2014 – TC Evan (Dec 2012) exacerbates damage incurred, with remaining standing riverine mangroves succumbing to windthrow; (D) July 2019 – Signs of recovery. Note that TC Winston also struck Ba in December 2016 yet there is little evidence of impacts. Imagery source: Google Earth Pro (2019).

Fig. 5. Clockwise from top left: (A) Healthy, open *Bruguiera gymnorrhiza* forest of Rewa $12_{\text{Coastal forest}}$. Evidence of selective logging was apparent at this site; (B) Example of the widespread damage and dead and downed wood which impedes regeneration in taller forest of the Ba Delta (Ba $1_{\text{TC damage}}$); (C) Panoramic image of cyclone damage on Yanuca Island (Ba $4_{\text{Island TC damage}}$) with healthy mangroves (Ba $3_{\text{Island intact}}$) in the background. Photograph credit: Clint Cameron (July 2019).

4. Discussion

4.1. Mangrove extent, landcover change and drivers of loss in Fiji

Our results revise the estimated extent of mangroves within the Fijian islands to 65,243 ha, an increase in spatial area of over 25,000 ha from the MFW database. It also revises estimates of mangrove loss between 2001 and 2018 to 1135 ha, with the vast majority of loss (~870 ha) occurring post-2012 and coinciding with TCs Evan and Winston. Historical mangrove loss in Fiji was estimated at 4313 ha between 1896 and 1986 (Lal, 1990), representing a decrease in spatial extent of approximately 6% at an annual rate of loss of 0.06% (extrapolating data on estimated historical extent from the current study). Combining historical (Lal, 1990) and contemporary (this study) datasets reveals that Fiji has lost an estimated 5447 ha of its mangrove ecosystems, or 7.7% of original extent since 1896. The historical loss of 4,313 ha of mangroves between 1896 and 1986 was driven primarily through conversion to sugarcane plantations (Lal, 1990), and in contemporary times this has been exceeded by losses resulting from tropical cyclones. This considerably changes our understanding of landcover dynamics in Fijian mangrove ecosystems and also indicates opportunities for restoration and conservation.

In the Pacific, the extent of mangrove loss in the 21st century ranges between 0% (Vanuatu) and 1.2% (77 ha, New Zealand. Hamilton and Casey, 2016). Net loss across all Pacific Islands is 8, 300 ha, or 1.29% of original extent at an annual rate of change of 0.06% between 1996 and 2016 (Worthington and Spalding, 2019). The spatial extent of contemporary mangrove loss in Fiji is the second highest recorded in the Pacific after Papua New Guinea (1763 ha) and slightly higher than that observed in Australia (1030 ha), which have lost 0.42% and 0.31% of mangroves respectively (Hamilton and Casey, 2016), although the timeframes over which mangrove loss is estimated varies between studies.

In recent decades, rates of global mangrove loss have declined. However, human activities continue to pose threats to mangroves in many countries. For example, conversion of mangroves to aquaculture ponds and oil palm plantations remains a significant driver of loss in some countries of South-east Asia (Hamilton and Casey, 2016, Richards and Friess 2016, Friess et al., 2020). Additionally, intensive harvesting of mangrove timber for fuelwood, conversion to rice (Friess et al., 2020) and changes to riverine hydrodynamics (e.g. reduced sediment and nutrient loads) through upstream freshwater extraction (Worthington and Spalding, 2019) have also been identified as key proximate threats to mangrove ecosystems. In contrast, our results highlight changes in both the temporal drivers of mangrove loss and the geographical variability of where loss occurs within Fiji, with anthropogenic stressors (e.g. conversion to sugarcane plantations) now superseded by natural disturbances from tropical cyclones.

4.2. Drivers of mangrove loss and implications for recovery

4.2.1. Spatial variation and patterns in mangrove loss from tropical cyclones

Tropical cyclones are recognized as significant drivers of change in mangrove ecosystems (Lugo, 2000). A recent review reported that 45% of losses in mangrove area caused by natural events globally was attributed to tropical cyclones (Sippo et al., 2018), ranking such impacts among the top non-anthropogenic disturbance (Krauss and Osland, 2019). This is certainly supported by our data for the Fijian Islands. However, there is considerable spatial variability in TC damage to mangroves with the south and west coasts of Viti Levu (including the capital, Suva, and the Rewa Delta) having lower vulnerability to the impacts of TCs compared to mangroves on the northern, north-eastern, and western coasts. An analysis of the 331 TCs in the South Pacific since 1970 (when satellite imagery became available) shows that TCs passed within ~50 km of Suva on only 6 occasions, with probably the most destructive being TCs Bebe (Category 3), Meli (Category 3), and Eric (Category 3) in October 1972, March 1979, and January 1985 respectively (APDRC n.d). In contrast, the Ba region has been struck on 9 occasions during the same period including 4 direct hits (TCs Gene, Mick, Evan, and Winston) within the last 9 years alone (Fig. 2).

Geographical variability in the vulnerability of Fiji's mangrove ecosystems to the impacts of TCs is also apparent within sites. TC Evan, for example, caused extensive damage to Tuva $6_{\text{Tall TC damage}}$ yet not the adjacent site of Tuva $5_{\text{Tall intact}}$, while just 14 km south there was no evidence of impacts within the Tuva Delta (Tuva$7_{\text{Intact lagoon}}$, Tuva 8_{Scrub}, Tuva $9_{\text{River margins}}$). It is possibly that the trajectory of TCs relative to coastline features causes highly localized spatial signatures of impact (Krauss and Osmond, 2019). For example, in the Philippines a single site (out of 7) had no visible effects from Super Typhoon Haiyan (November 2013) despite all sites being in close proximity to one another (Primavera et al., 2016). Additionally, different forest types are more susceptible to TC damage than others. Saenger (2013) suggested that individual trees susceptible to windthrow are likely to be those with weakly developed cable root systems, or those with structures weakened by erosion, slumping or biological agents (Krauss and Osmond, 2019). In Florida, taller trees that had been fertilized were more susceptible to hurricane damage than scrub trees (Feller et al., 2015). Whether such factors influenced the susceptibility of individual Rhizophora spp. trees within Tuva $5_{\text{Tall TC damage}}$ rather than trees in adjacent plots (Tuva $6_{\text{Tall intact}}$) requires further research.

4.2.2. Impacts of tropical cyclones on mangrove forest structure

While the extent of initial damage from TCs is determined by local storm intensity and pathway, species composition and tree height influence the degree and relative severity of damage (Feller et al., 2015, Asbridge et al., 2018). Proximate factors driving direct damage from TCs include sustained high winds resulting in crown damage (defoliation and snapping of branches), windthrow and bole damage (Asbridge et al., 2018; Kjerfve, 1990; Stocker, 1976), while strong waves and storm surges can change hydrology and sediment distribution as a function of erosion and/or accretion events (Cahoon and Hensel 2002). Additionally, the deposition of downed wood can alter hydrological regimes as well as localized soil chemistry which may impede propagule establishment and regrowth (Asbridge et al.. 2018). These factors have important implications for carbon storage in mangroves, subsequent recovery and the on-going, long-term sustainability of mangrove ecosystems affected by TCs.

The extensive damage to predominantly taller mangroves (e.g. Ba $1_{\text{TC damage}}$, Tuva $5_{\text{TC Damage}}$, Ba $4_{\text{Island TC damage}}$, and Ra $13_{\text{Hinterland}}$) can primarily be attributed to greater periods of exposure to higher winds and strong gusts which cause windthrow and defoliation. Such impacts have also been observed in Australia (Feller et al., 2015, Asbridge et al., 2018), Nicaragua (Roth, 1992), Myanmar (Aung et al., 2013) and the Philippines (Salmo III and Gianan, 2019; Villamayor et al., 2016). Additionally, there is evidence that mangrove species in the family Rhizophoraceae (e.g. Rhizophora spp. and Bruguiera spp.) are more susceptible to the impacts of TCs than other mangrove species as they cannot resprout via remaining plant material or coppicing (regrowth of stumps) after damage, and are dependent on recruitment of juveniles to regenerate. In contrast, species within families such as Avicenniaceae readily resprout via coppicing (Aung et al., 2013, Woodroffe and Grime, 1999, Krauss and Osland, 2019). For Viti Levu Bay, relative tree height is likely the main factor influencing the extent of damage in forest assemblages given both the taller interior, Bruguiera gymnorrhiza dominated forest of Ra $13_{\text{Hinterland}}$ and shorter in statute, coastal fringing Rhizophora spp. composed Ra $14_{\text{Coastal margin}}$ are within the Rhizophoraceae family. Structural damage was not apparent in Ra $14_{\text{Coastal margin}}$, although this site suffered extensive defoliation in the immediate aftermath of TC Winston (Alivereti Naikatini *pers. comm.* 2019). Most large branches remain intact at this site, and this probably facilitated epicormic foliage recovery in contrast to Ra $13_{\text{Hinterland}}$ where tree structural damage was severe. For Ba $1_{\text{TC damage}}$, the few remaining tall standing live trees showed signs of epicormic foliage recovery where crown structure and large branches remained intact, with the sprouting of new leaves occurring from buds located in terminal branches of mature *Rhizophora* spp. (Bardsley, 1985).

In contrast, the scrub mangroves of the Ba Delta (Ba 2_{scrub}) as well as trees in canopy gaps were not as susceptible as taller vegetation to the impacts from sustained high winds and did not suffer the same level of mortality. Trees that are smaller in stature can be more resistant to structural effects given a lower position within the vertical wind boundary layer, or perhaps submergence during storm tides, which facilitates some protection from strong winds (Smith et al., 1994; Krauss and Osmond, 2019). However, while mortality was minimal and there were no changes in forest extent or cover at Ba 2_{Scrub}, the legacy of TC impacts was still apparent with the mass of fine downed wood equivalent to that of above ground living biomass. Both the carbon stock and percentage of downed wood within the above ground carbon pool at Ba 2_{Scrub} were much higher than scrub mangroves in other parts of the world (DDWD = 1.2 –1.8 Mg C ha^{-1} and 2.3% - 5.6% of AGB; Kauffman and Bhomia, 2017 and Ochoa-Gómeza et al., 2019 respectively) and similar to a site impacted through storms and human disturbances in Mexico (DDWD = 3.4 Mg C ha^{-1} and 25% of total AGB; Ochoa-Gómeza et al., 2019). Although damage to the mangroves from TCs was evident, TCs may also have positive biotic feedback effects particularly in hypersaline and/or nutrient-poor environments which support scrub mangroves (Krauss and Osmond, 2019). Excessive sedimentation during TCs can bury roots and lead to mortality (Ellison, 1998; Paling et al., 2008), but moderate sediment deposition can stimulate plant growth and give rise to gains in soil surface elevation as storm surges deliver nutrient rich sediments and freshwater inputs (Krauss and Oslond, 2019, Lovelock et al., 2011). The mass of fine downed wood caused by TC Evan at Ba may enrich soil layers as it decomposes and could also contribute to higher soil carbon and nitrogen concentrations observed at this site (Cameron et al., 2020), but whether the scrub mangroves of the Ba Delta have benefited from nutrient enrichment through sedimentation associated with storm surge requires further study.

TCs can also cause localized changes in soil chemistry, hydrology, and the productivity of mangroves, which can all influence recovery. Firstly, the decomposition of downed organic matter can result in anoxic sediments (low dissolved oxygen) which can create unfavorable conditions for propagule recolonization and establishment (Cahoon et al., 2003; Mendelssohn et al., 1995) and may be one reason for the lack of advance recruitment observed in cyclone damaged forests of Ba $1_{\text{TC damage}}$, Ba $4_{\text{Island TC damage}}$, and Ra $13_{\text{Hinterland}}$. Additionally, storm surge or strong rainfall runoff from TC Winston may have led to mortality of seedlings from the understory in these sites due to prolonged inundation as occurred in some mangrove forests after Cyclone Eline in Mozambique (Macamo et al., 2016). Hydrological modifications from blockages to tidal drainage or ebb flows caused by downed wood and sedimentation either pre- or post-TC Winston may also be preventing further regeneration or recruitment of propagules (Lewis et al., 2016; Krauss and Osmond, 2019). For instance, on Yanuca Island (Ba $4_{\text{TC damage}}$), it is likely that the sheer mass and scale of downed wood formed a physical barrier preventing the dispersal and inland penetration of propagules on high tide, which is similar to observations made in damaged mangrove areas in Indonesia (Cameron et al., 2018). Hydrological modification (e.g. blockages from downed wood) is also likely to be a key factor controlling recovery in Viti Levu Bay where seedlings and saplings of *Bruguiera gymnorrhiza* were absent from interior zones, probably because of changes in freshwater influx. Finally, propagule production from the remaining healthy mangroves of the Ba Delta, Yanuca Island and Viti Levu Bay may have been supressed due to reduced productivity and propagule production of remaining maternal trees (Krauss and Osmond, 2019).

Importantly for Fiji, the legacy of past tropical cyclone disturbance followed by subsequent recovery often defines the current mangrove ecosystem state (Krauss and Osmond, 2019). For instance, while TC impacts in Ba $1_{\text{TC damage}}$ are extensive, this forest also shows signs of recovery in an emergent cohort of living trees which display relatively uniform structural characteristics (e.g. 88.3% of all living trees had a DBH <10 cm and of those, a median DBH of 4.2 cm). The size classes of these trees were similar to those with an average age of ~5–6 years in Rhizophora spp. dominated forests of the Mekong Delta, Vietnam (Phan et al., 2019). Additionally, the impacts from TC Evan probably influenced the severity and degree of impact from TC Winston. As illustrated in Fig. 4, TC Winston had few impacts (e.g. defoliation or structural damage) on the developing mangrove forest despite this event being the strongest of the cyclones to hit the Ba Delta. This is probably related to the fact that trees were only ~3–4 years old when TC Winston struck and thus less susceptible to windthrow than taller, more mature trees. Understanding the biophysical conditions imposed by a legacy of past TC disturbances are paramount to projections of the impacts of TCs on mangroves and their eventual recovery.

4.2.3. Selective harvesting and small-scale clearance

While Rewa and Tailevu exhibited less TC damage than other regions of Fiji, there was clear evidence of small scale, selective harvesting. Selective harvesting is a management practice whereby only certain individuals from a given stand are removed, leaving other trees standing (Pommerening and Murphy, 2004). This serves to maintain the integrity of a forest's overall coverage, although potentially may also alter biodiversity and ecosystem function (Dadouh-Guebes et al., 2005). Small-scale harvesting is recognized as one of the most widespread forms of resource use in mangrove forests worldwide (Scales and Friess, 2019) and, unlike larger-scale forest clearance (e.g. clearing for agriculture, or the impacts of dredge spoil disposal observed in other parts of the Rewa and Ba Delta's), it is not often detectable through remote sensing analysis (Dahdouh-Guebas et al., 2005; Scales and Friess, 2019). The small-scale harvesting of mangroves by indigenous communities around the world remains poorly understood (Alongi and de Carvalho, 2008), apart from a few studies in (mostly) terrestrial biomes which suggest this practice can have significant and often cumulative impacts on forest structure, composition and regeneration (Luaga et al., 2004; Ticktin, 2004). For instance, in the Philippines small-scale mangrove harvesting led to a 70–95% decline in the density of trees in the 5–15 cm DBH size class compared with uncut stands (Eusebio, 1986; Walters, 2005a). Walters (2005a) also observed a three-fold increase in canopy openness as well as a decline in basal area and mean tree DBH, although in Timor Leste there was little demographic evidence of a significant change in species composition with small-scale harvesting (Alongi and de Carvalho, 2008).

In the selectively harvested Rewa forests assessed in this study (Rewa $11_{\text{Hinterland}}$ and Rewa $12_{\text{Coastal forest}}$), such impacts were not readily apparent. The low harvesting pressure and ratio of stumps to living trees (7.8% and 2.9% of trees extracted per ha^{-1} and 0.0078:1 and 0.0014:1 of stumps to trees for Rewa $11_{\text{Hinterland}}$ and Rewa$12_{\text{Coastal forest}}$) is lower than mean harvesting rates of 7–50% observed from small-scale harvesting of mangroves in Madagascar, the Philippines, Timor-Leste, Venezuela and South Africa (Scales and Friess, 2019; Walters, 2005b; Alongi and de Carvalho, 2008; Lo´pez-Hoffman et al., 2006; Rakaran et al., 2004). The low level of extraction may even mimic natural rates of mortality reported in Indonesia (M. Sillanpaeae pers. comm. 2019) or the effects of minor natural disturbances such as lightning strikes (Hauff et al., 2006). The comparatively low harvesting pressure (noting that we did not collect data on the *rate* of extraction) and high seedling and sapling densities recorded, which may reflect high light levels in the understory enhanced by small canopy gaps, corresponds to overall healthy, intact, and structurally complex forests.

The method employed for extraction can also impact the long-term health of forests. Within Rewa's mangrove forests, stems of predominantly *Bruguiera gymnorrhiza* are extracted using chainsaws from hinterland or interior zones, rather than *Rhizophora spp.* from coastal or river margin zones. This can limit impacts on the structural integrity of forests by reducing edge effects such as windthrow on interior trees or erosion of mangrove substrate caused by the removal of trees at a forest's coastal periphery. Mechanical removal (e.g. heavy diggers with a shovel or rake attachment), by difference, can significantly impair mangrove recruitment through soil compaction as occurred in New Zealand (Horstman et al., 2018). In contrast, logs within Rewa $11_{\text{Hinterland}}$ are cut to size, manually transported to river margins, and then taken by longboat to local villages, while within Rewa $12_{\text{Coatsal forest}}$ logs are moved around 100–300 m to landward terrestrial margins (Eliki Senivasa *pers. obser.*). These (probable) low impact methods of removal ensure mangrove soils remain intact and, when combined with the low numbers of trees removed, can be significant factors in maintaining forest health.

The extent and effect of selective harvesting and small-scale clearance on mangroves in Fiji should be further assessed in the future, particularly given our field surveys were limited in spatial extent. For example, at Rewa $11_{\text{Hinterland}}$ we saw visual evidence of clear-cutting where only a few scattered living trees remained. This area was not detected during the analysis of mangrove cover loss and signals a need for further investigation of the prevalence, distribution, rate (i.e. no. trees extracted per year) and impacts from extraction – whether small scale selective harvesting or clear felling – within other areas of the Rewa Delta not assessed in this study. Although the observed selective harvesting pressure (i.e. within plots) and the extent of clear-felled areas appear low, the spatial extent and potentially rate of extractive use of mangroves may well be greater than our current understanding.

4.3. Potential opportunities for mitigating climate change and improving management of mangrove landscapes

The differing drivers of landcover change coupled with the relative spatial extent of mangrove forests suggest a number of options for implementing management activities which may contribute to Fiji's climate change mitigation and adaptation targets. For instance, activities that reduce CO_2 emissions or sequester carbon could lead to the development of blue carbon projects (Table 3). For forest carbon projects,

Table 3
Potential interventions for climate change mitigation or blue carbon projects for Fiji.

Intervention types	Potential locations	Description
Afforestation, reforestation, revegetation (ARR)	Ba Delta, Yanuca Island, Viti Levu Bay	ARR is an eligible project category under the Verified Carbon Standard, combining some or all of the three elements of afforestation, reforestation and revegetation. It covers activities that increase carbon stocks in woody biomass (and in some cases soils) by establishing, increasing and/or restoring vegetative cover through the planting, sowing and/or human-assisted natural regeneration of woody vegetation (The REDD Desk, 2020). Afforestation involves establishing vegetative cover on lands that were not previously vegetated. In the context of blue carbon in Fiji, for instance, ARR projects could involve the restoration of mangroves degraded by activities such as agriculture (e.g. conversion to sugarcane), clear-felling, dredge spoil placement, or damaged by tropical cyclones (i.e. reforestation or revegetation).
Avoided deforestation	Rewa Delta and southern coastline of Viti Levu	This project type includes activities that reduce net GHG emissions by stopping or reducing planned or unplanned deforestation or degradation on forest lands (The REDD Desk, 2020). For instance, this might involve cancelling a clear-felling logging concession for planned activities on state land, or working with communities to reduce the level of ad-hoc logging for unplanned activities.
Improved forest management	Rewa Delta and southern coastline of Viti Levu	Forest management activities which result in increased carbon stocks within forests and/or reduce greenhouse gas emissions from forestry activities when compared to business-as-usual forestry practices (The REDD Desk, 2020). For instance, this might entail shifting from clear-felling of forests to selective logging which would result in net carbon gains.

management interventions that develop certifiable carbon offsets need to pass a test of 'additionality' to determine whether an emissions reduction or removal would have occurred in the absence of the intervention (The REDD Desk, 2020) and therefore is not a continuation of 'business as usual'. For example, under an avoided deforestation scenario there needs to be a specific driver of deforestation (e.g. logging) which can be alleviated to avoid on-going emissions. In the case of reforestation, interventions must increase CO_2 capture through regrowth above what would normally occur. If there is no driver of loss to avoid and forests remain largely intact as in Nadroga-Navosa where there is little evidence of either historical or on-going mangrove loss (e.g. from harvesting), projects would be unable to meet this requirement.

4.3.1. *Assisted regeneration of mangroves impacted by tropical cyclones*

Of the potential management interventions listed in Table 3, the damage incurred from TCs within the Ba Delta, Yanuca Island and Viti Levu Bay may present an opportunity to develop interventions focused on augmented recovery under an 'Afforestation, Reforestation, Revegetation' (ARR) Verified Carbon Standard (VCS) framework. The loss of mangrove cover within the taller riverine mangroves of Ba $1_{\text{TC damage}}$ could potentially constitute a moderate scale (~168 ha) opportunity for assisting with the process of natural recovery. While some revegetation of this forest type is already apparent in an emergent cohort of 5–6 year old trees, there appears to be limitations to recovery possibly because of (a) a lack of suitable physical recruitment space due to high levels of downed wood; (b) downed wood altering soil chemistry; (c) reductions in seedlings and saplings post-TC Winston; and (d) limitations to seedling production and dispersal because of reduced hydrological connectivity. Further site-specific research could seek to investigate the biophysical conditions imposed by a legacy of past TC disturbances which may be hindering mangrove recruitment. For example, the physical removal of coarse woody debris may alleviate issues around space for seedlings as well as reducing localized impacts on soil chemistry caused from decomposition, provided this can be achieved without compacting soils. Moreover, the clearing of tidal creeks could improve hydrological connectivity which is necessary for the dispersal of propagules from healthy mangrove estate and could drain water logged soils.

Yanuca Island (Ba $4_{\text{Island TC damage}}$) may also present a potential opportunity to undertake a spatially compact, small-scale reforestation project given there is little indication of recovery at this site. The organic soils of this site are likely substantial sources of CO_2 as they decompose (Cahoon et al., 2003), with this affect not currently offset by sequestration given the lack of living trees or apparent regrowth. Additionally, soil material from the cyclone impacted area are leaching out and negatively affecting adjacent seagrass beds. While dead tree roots and stumps can be effective at consolidating or holding soils in place in the short- to medium-term (Murray et al., 2011), the lack of fine root turnover and leaf litter deposition from living trees precludes the accumulation of new autochthonous soil organic matter to replace lost soil. The lateral displacement of soils combined with the ongoing decomposition of remaining organic matter may lead to soil subsidence and compaction which could eventually lead to soil elevations no longer suitable for mangrove growth (Asbridge et al., 2018; Lang'at et al., 2017, Cahoon et al., 2003), further increasing the rationale for attempting restoration at Yanuca Island. Moreover, the island is also important for mud crab (*Scylla serrata*) harvesting and thus restoration could have biodiversity and community benefits which may enable accreditation in schemes such as the Climate, Community and Biodiversity Standards (Cameron et al., 2019).

The relatively large size (~307 ha) and contiguous area of *Bruguiera gymnorrhiza* mangrove forest damaged by TC Winston in Viti Levu Bay (Ra $13_{\text{Hinterland}}$) which is not recovering also makes this a potential option for an augmented or assisted recovery (ARR) blue carbon project. Like the Ba Delta, further research is required in order to assess the biophysical factors limiting recovery which would help to determine the feasibility of interventions.

Finally, given the long-term recovery of mangroves is often dependent upon the restoration of hydrological regimes as well as both the frequency, intensity and disturbance legacies of TCs (Asbridge et al., 2018; Krauss and Osmond, 2019), successful interventions would need to be framed against the degree of risk of future reoccurrence undoing carbon gains ('permanence'). While the frequency of tropical cyclones is expected to decline in the Western Pacific in response to climate change (Knutson et al., 2015), an increase in the frequency of the most intense storms (e.g. TC Winston) and the amount of rainfall produced combined with an increased poleward expansion in the range of tropical cyclones (Krauss and Osmond, 2019) creates significant uncertainty and risk for mangrove restoration projects in Fiji which needs to be considered further.

4.3.2. *Avoided deforestation and / or improved forest management*

The Rewa Delta is Fiji's largest contiguous area of mangroves and is therefore a recognized priority site for conservation by the Fijian government (MoE, 2018). Our data shows that while there is some extractive harvesting pressure within Rewa, it is small scale and localized in extent. The management of mangroves within the Rewa Delta is regulated through a moratorium enacted in 2013 which prohibits commercial logging but allows for subsistence extraction by local communities, with predominately *Bruguiera gymnorrhiza* extracted for use as timber in community housing and squatter settlements (Conservation International, 2018). Research has also shown that governance decentralization and community management can improve mangrove condition if strong community institutions are present to enforcement common rules of natural resource management (Osmond and Friess, 2019). Our initial results from Rewa, particularly from Rewa $12_{\text{Coastal forest}}$, would appear to support these findings, with mangrove utility regulated and enforced most strongly at the community level even though the Fijian Government's Ministry of Forestry maintains ownership and extractive permit rights for the country's mangrove estate (Alivereti Naikatini pers. comm. 2019). While extraction is localized and small-scale, there are concerns that an increased influx of people migrating from inland rural areas to coastal communities will subsequently drive demand for construction materials needed to build new houses (Conservation International, 2018). For instance, the 2017 Fiji Bureau of Statistics census reports an increase in the proportion of people residing in urban settlements from 37.2 per cent in 1976 to 55.9 per cent in 2017. Additionally, *Bruguiera gymnorrhiza* is highly valued as a fuelwood for use in traditional Hindu crematorium ceremonies given its high calorific content and density. Parts of the Rewa Delta remain subject to extraction for this end use, particularly where mangroves are situated in close proximity to main roads which enables an easier transportation of wood (Conservation International, 2018).

While harvesting pressure may be low, evidence of extraction in almost all sites assessed indicates that impacts may be more wide-spread and operate at larger spatial scales than our current understanding suggests, particularly given remote sensing analyses were unable to identify small-scale disturbances. Assessing the scale and degree of clearance would require further field research, including conducting semi-structured interviews with communities that reside within and around Rewa to complement existing data. If a significant spatial scale of extraction was detected, then this may present a viable opportunity for avoided deforestation or improved forest management projects by increasing the spatial area across which forest management could be implemented.

5. Conclusions

The rate and proportional loss of mangroves in Fiji in the 21st century is comparable to other countries around the world where mangroves occur. The limited spatial extent of contemporary loss (1135 ha over 18 years) compared to historic losses suggests large-scale avoided deforestation or restoration projects are not particularly appropriate, although smaller scale projects may be more feasible. For instance, of the few mangrove projects registered with the Verified Carbon Standard, the spatial extent ranges from 4624 hectares to 10,415 hectares and all involve the large-scale replanting of deforested or degraded mangrove areas (Cameron et al., 2019). Sea-level rise, however, may eventually render some parts of the significant historical area of mangroves converted to sugarcane (4, 313 ha) untenable through salinization, and this could facilitate larger scale restoration projects.

Of importance, the high carbon sequestration potential of mangroves coupled with the ability to mitigate CO_2 emissions from soils of damaged forests in comparison to other habitat types may suggest that the *magnitude* of greenhouse gas mitigation benefits may offset to some degree the *scale* (i.e. spatial extent in hectares) of potential interventions. Therefore, for Fiji even small-scale (~20 ha) rehabilitation projects may be feasible in terms of the magnitude of carbon offsets able to be generated, particularly when other ecosystem services such as coastal protection, fisheries, cultural and biodiversity values are considered.

In summary, for Fiji the relevance of establishing blue carbon interventions (e.g. ARR, avoided deforestation or improved forest management) at particular sites is informed both by the drivers of loss as well as the availability of current methodologies to account for these differing scenarios (Mack et al., 2012; Wylie et al., 2013; Needleman et al. 2019). Our results suggest a need to extend existing methodologies to encompass a wider range of intervention approaches. This has relevance not only for Fiji but other small island states throughout the Pacific, as well as regions facing similar impacts from tropical storms such as the Caribbean.

Declaration of Interest Statement

The authors declare that they have no conflict of interest with regards to the manuscript '*Landcover change in mangroves of Fiji: Implications for coastal climate change mitigation and adaptation*'.

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Supplementary material

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