



Modeling Catchment-Scale Nitrogen Losses Across a Land-Use Gradient in the Subtropics

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Changing land use in subtropical and tropical catchments to farmland can result in higher nitrogen (N) loss to aquatic ecosystems. Here, we developed a lumped water and N balance model to estimate regional N losses to creeks at catchment scale within understudied subtropical catchments in Australia. The conceptual water balance model CoCa-RFSGD was extended by the nitrogen mass balance in top and subsoil by adding nitrogen cycle transformation estimates depending on meteorological, soil, and land-use properties. The model estimates the impact of pristine and agricultural land use on catchment-wide water quality using only low-order creek samples as water quality measurements of nitrate and nitrite (NO_x) with increased model performance with increased agricultural coverage. The model revealed that an agricultural proportion of 3% in the study site drove a 3.5-fold increase of N losses to creeks and a 6.7-fold increase of N losses to the atmosphere compared to catchments without agriculture. Agricultural land use lost 92 kg-N ha⁻¹, 85% of which evaded to the atmosphere and 15% was discharged via surface waters. A change from forest to cleared land may increase the total denitrification potential of a catchment. Overall, our lumped model provides a simple but effective tool to upscale local aquatic water quality measurements to the catchment scale, allowing for assessment of changing land use on aquatic N loads in areas with limited data availability.

Keywords: modeling soil nitrogen cycling, subtropics, horticulture, nitrate leakage, eutrophication

1. INTRODUCTION

Humans started to alter the nitrogen (N) cycle centuries ago by cultivating N₂-fixating legumes (Smith and Nesbitt, 1995), burning wood and coal (Smil, 1994), and developing industrial fertilizers (Smil, 2004). While the fate of much of this additional N depends on local conditions (Leuzinger et al., 2011; Dieleman et al., 2012), large amounts of this excess anthropogenic N discharged to water bodies or the atmosphere, which can effect human and environmental health (Townsend et al., 2003; WHO, 2003). Excess N in aquatic systems can increase greenhouse gas emissions (IPCC, 2013), change biological communities (Diaz and Rosenberg, 2008; McCrackin et al., 2017), and drive eutrophication (Rabalais, 2002; Howarth and Marino, 2006).

A large portion of the additionally available reactive N originates from fertilization and manure application (mainly as urea and ammonium) on agricultural fields, which microbial soil communities transform by hydrolysis, nitrification and denitrification to ammonium, nitrate and

atmospheric N, respectively (Singh et al., 2010). Microbial N processes can drive N losses by either nitrate (NO_3) and ammonium (NH_4) leaching into groundwater and surface water or atmospheric N losses as dinitrogen (N_2) (Mosier et al., 1998; Beaulieu et al., 2011) and nitrous oxide (N_2O) (Wang et al., 2018). Local climate and soil conditions can influence microbial processes; for example, temperate climates have a higher rate of fertilizer N discharging into groundwater and surface water bodies, whilst the tropics may experience higher N losses into the atmosphere because denitrification increases with temperature and soil moisture (Talling et al., 1998; Houlton et al., 2006; Lee et al., 2019).

Contrary to most temperate regions in the Northern Hemisphere, many Southern Hemisphere subtropical and tropical regions still have pristine and arable land available, and this land is being rapidly transformed into agriculture (FAO, 2001; Lee et al., 2019). The humid tropics in particular have experienced a transformation of the rainforest (55% intact forest and 28% disturbed forest) into agricultural land (Lambin, 1997; Gibbs et al., 2010). Land use change from forest to agriculture affects the nitrogen cycle in multiple ways: (1) land use changes can drive localized climate extremes, such as increased heat days and breaks of monsoon, affecting soil microbial activities (Paeth et al., 2009); (2) there is an increase in greenhouse gas emissions, such as N_2O from increased manure and fertilizer application (Davidson, 2009); (3) there can be lower nitrogen sequestration in agricultural soil than in forest soil (Lee et al., 2019); and (4) there may be additional losses of reactive nitrogen into ground and surface waters as well as losses of N_2 to the atmosphere (Shen et al., 2008).

Land use and resulting N cycling changes may affect tropical climates more intensely. Although ~90% of global N fertilizers are applied in Asia, Europe, and North America, the tropics are expected to transmit comparably larger amounts of N into the atmosphere and surface water bodies (Galloway et al., 2004; Hietz et al., 2011; Lee et al., 2019). Tropical regions have an increased source of reactive N through an increased rate of lightning, as well as biological N fixation in rainforests via N-fixing canopy legumes (Vitousek et al., 2002; Cleveland et al., 2013). While temperate regions experienced an intensive land use change in the early 20th century, the soil nitrogen cycle in these regions has begun to stabilize, resulting in higher N sequestration potential (Lee et al., 2019).

Several modeling tools have been developed using transient mass-balances with empirical equations to assess the impact of land use and climate on soil N cycling and microbial N transformations (Lin et al., 2000). It is generally preferred to build models to suit specific study locations. A total of 80% of all distributed models simulating catchment N losses between 1992 and 2010 use one of the five most common models (Wellen et al., 2015). These models often include the soil interaction with lakes and the hydrology of snow at temporal resolutions <1 day as well as complex processes, such as crop growth, the soil carbon cycle, and the energy balance [see models DNDC (Liu et al., 2006), SWAT (Lam et al., 2010), and DAYCENT (Parton et al., 1998)]. Modeling small-scale processes at large scales demands high spatial and temporal input data regarding climate, soil, aquifer, fertilization, and land use. These data are often

not available for tropical and subtropical areas that are rapidly changing. Water quality measurements and nitrogen loads are more abundantly available.

Here, we developed a model based on an conceptual understanding of the soil nitrogen cycle to estimate (1) catchment N surplus at data-poor regions using NO_x creek water measurements only and to estimate (2) the impact of local land use changes in the subtropics. The model is an extension of the water balance model CoCa-RFSGD (Hajati et al., 2019). It was designed to simulate data-poor catchments in Indonesia because it relays on data sets, which are globally available. The model was applied in subtropical creeks of northern New South Wales (NSW), Australia.

2. STUDY SITE

Our model and observations were performed on the east coast of Australia extending from the town of Bellingen in the South ($30^\circ 27' 8.07''\text{S}$, $152^\circ 53' 52.67''\text{E}$) to Yamba in the North ($29^\circ 25' 59.99''\text{S}$, $153^\circ 20' 59.99''\text{E}$). We modeled catchments draining east from the Great Dividing Range (GDR) into the Pacific Ocean (Figure 1A). The study area has 217 sub-catchments with similar climate, stream order and geomorphology but a different percentage of agricultural, cleared land, and forest cover, allowing us to study the influence of land use on catchment-wide N losses.

Precipitation within the study area is drained through a network of small rivers and creeks into the ocean (Figure 1B). We covered a narrow area of 900 km^2 along the coastline in the humid subtropics. Coffs Harbour, the largest city of the study area, is located in the South ($30^\circ 18' 8.21''\text{S}$, $153^\circ 07' 8.03''\text{E}$) and has an annual rainfall of 1,700 mm with ~60% of precipitation occurring from January to May (BOM, 2019). The average daily temperatures range between 14 and 23°C . Soils are generally strongly acidic with associated aluminum toxicity, high topsoil organic matter, low subsoil permeability, and low fertility (OEHL, 2018), which is typical for tropical soils. The aquifer is composed of fractured Palaeozoic rocks or Mesozoic intrusives, forming an aquifer with a local flow system in the South but local and intermediate flow systems in the North (BOM, 2012). Hence, the region has small catchment sizes, and precipitation that recharged the aquifer flows for only short distances before discharging into local creeks.

Land use changes from horticulture dominated in the South, protected forests being found in the North, and some urbanized areas in between. The dominant agricultural land use of the study area has changed from banana plantations to blueberry horticulture since the year 2000 (Conrad et al., 2019). This land use change trend is expected to continue (Bevan, 2006; Government, 2017). The acidic and infertile soil of this area is ideal for blueberry farming (Hanson, 2006). NSW grows >90% of Australia's blueberries, generating annually ~AU\$250 million per annum, with most of the industry concentrated within our study region (RDA Northern Rivers, 2017). The steadily growing industry raises increasing concerns for creek water quality (White et al., 2018). Creek waters draining blueberry farms show significantly higher NO_x concentrations than forested catchments, with an estimated loss of ~20% of applied N

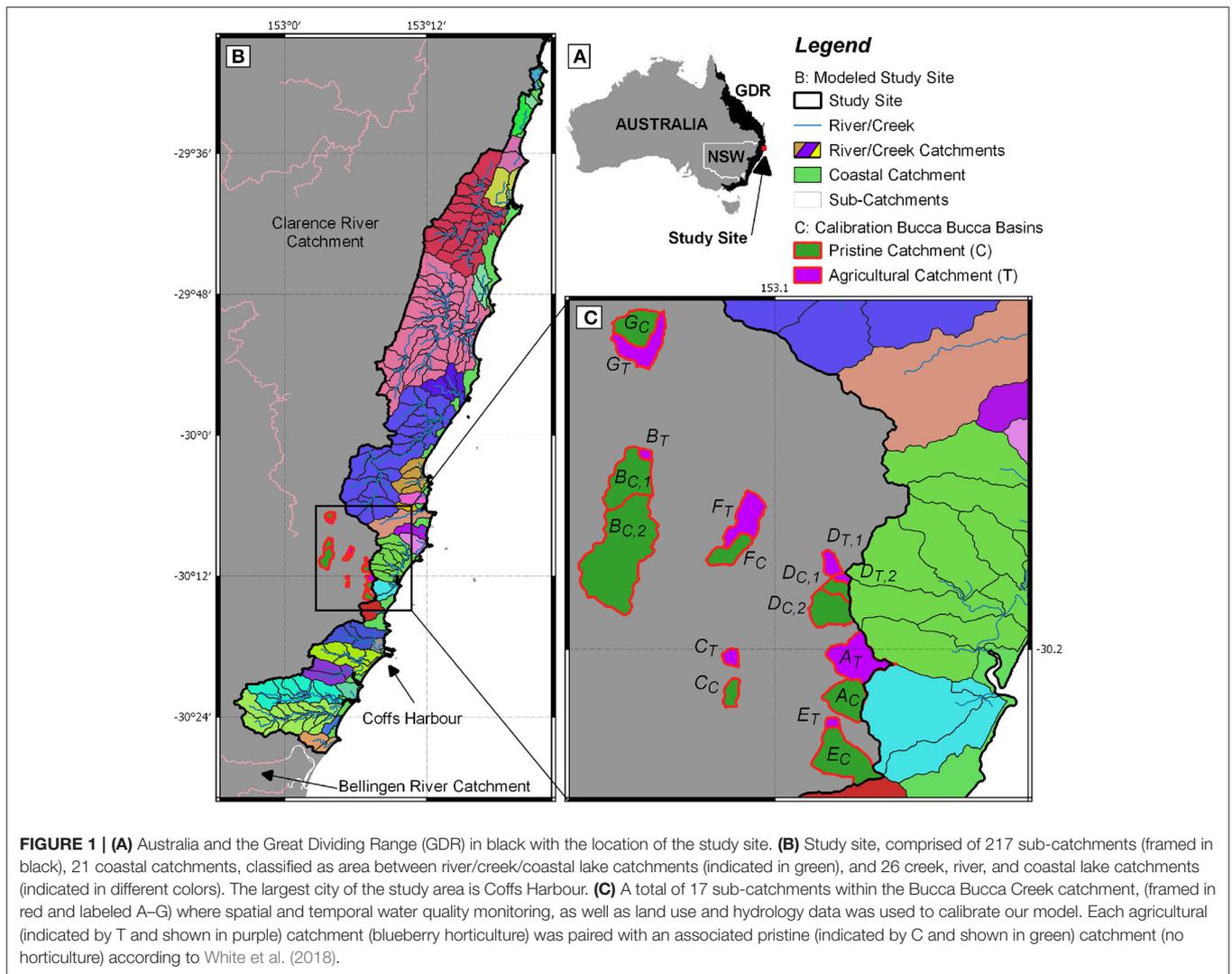


FIGURE 1 | (A) Australia and the Great Dividing Range (GDR) in black with the location of the study site. (B) Study site, comprised of 217 sub-catchments (framed in black), 21 coastal catchments, classified as area between river/creek/coastal lake catchments (indicated in green), and 26 creek, river, and coastal lake catchments (indicated in different colors). The largest city of the study area is Coffs Harbour. (C) A total of 17 sub-catchments within the Bucca Bucca Creek catchment, (framed in red and labeled A–G) where spatial and temporal water quality monitoring, as well as land use and hydrology data was used to calibrate our model. Each agricultural (indicated by T and shown in purple) catchment (blueberry horticulture) was paired with an associated pristine (indicated by C and shown in green) catchment (no horticulture) according to White et al. (2018).

fertilizer (White et al., 2018). The recommended annual fertilizer application on blueberries is 121 kg-N ha^{-1} and 83 kg-P ha^{-1} (Doughty et al., 1988; Ireland and Wilk, 2006), though actual application rates are not reported (Conrad et al., 2019).

3. METHODS

3.1. Model Conceptualization

To quantify the impact of a land use shift from pristine forest to blueberry farming in the subtropics, we combined the catchment water balance model CoCa-RFSGD (Hajati et al., 2019) with N mass balances based on well-established N equilibrium equations and first-order kinetics. This enables us to carefully draw conclusions on atmospheric losses, while the model is only calibrated on aqueous N losses. If the N cycle is closed, N input equals N output without losses to surface water bodies. Catchment N input occurs through biological N fixation (BNF), atmospheric N deposition, fertilizer application and plant litter fall. Catchment N output occurs through

harvest, grazing animals, plant uptake and microbiologically through volatilization and denitrification. The N cycle is open when N inputs become larger than soil microbial bacteria can consume and plants can take up; microbial atmospheric losses via volatilization and denitrification will thus increase, and excess N will leach into ground and surface waters. We will further refer to additional reactive N caused by an open N cycle as nitrogen surplus (N_{surpl}).

We focus on catchment N_{surpl} entering surface waters and atmosphere, since the integrated catchment output of N_{surpl} in surface water is related to catchment N_{surpl} . This aspect is particularly relevant for regions where fertilizer application rates are not known and where, given governments' fertilization application, guidelines function as recommendations rather than regulations. Because the streams are groundwater-fed and located near the headwaters of the catchment, we assume that nitrogen loads in the low order stream of the study region represent the integrated catchment aqueous N loss. There is no trans-catchment groundwater pumping in the region.

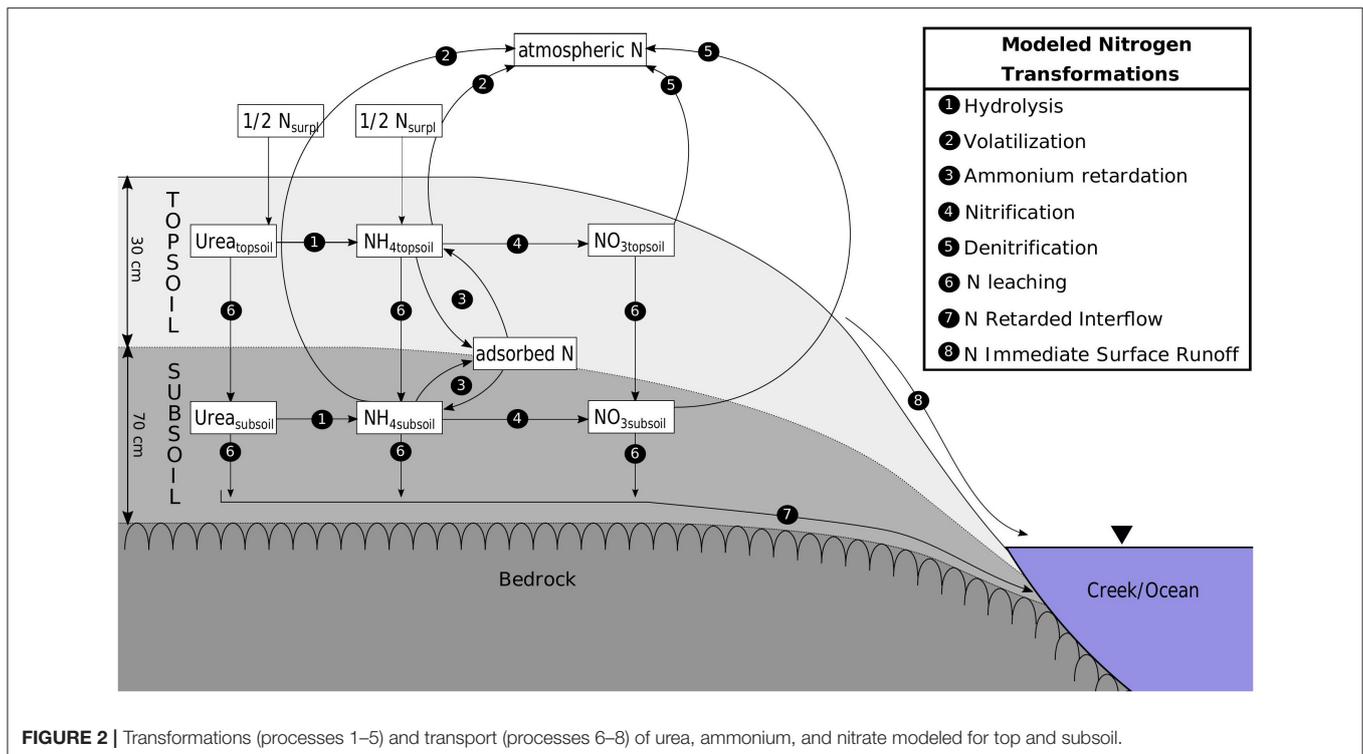


FIGURE 2 | Transformations (processes 1–5) and transport (processes 6–8) of urea, ammonium, and nitrate modeled for top and subsoil.

The catchment N_{surpl} leaching to creeks and emitting to the atmosphere is dependent on natural and anthropogenic N input in the catchment, soil properties (pH, sand, silt, clay content, and bulk density) and meteorological properties (rainfall, evapotranspiration, soil moisture, and soil temperature).

The CoCa-RFSGD model (Hajati et al., 2019) is a water balance model working at catchment scale with a daily temporal resolution calculating topsoil and subsoil water content based on soil water balances. The model uses two unsaturated layers, lumped 30 cm topsoil (root zone), lumped 70 cm subsoil layer, and an aquifer layer with unspecified depth. Water input through precipitation surplus (precipitation minus evapotranspiration) fills the topsoil until the soil is saturated, and additional water is then transported through surface water runoff to surface waters. Leakage from the topsoil to subsoil and from subsoil to aquifer as well as capillary rise from subsoil to topsoil are calculated using unsaturated hydraulic conductivities calculated using the Van Genuchten (1980) approach. The study site is located on fractured rock, which is described to mainly form local flow systems (BOM, 2012); creek water is thus assumed to contain surface runoff and retarded subsoil runoff (interflow) and a groundwater layer is neglected (process 7 and 8 in **Figure 2**). Radon measurements at the Bucca Bucca creeks and a coastal catchment within our study area also suggest groundwater discharge is a negligible source of N (Conrad et al., 2018; White et al., 2018). We extended this model by mass transport and transformation of urea, NH_4 , and NO_x .

We calibrated the model using weekly NO_x measurements from February to May 2017, while the model period is 9 years

from 2009 till 2018. In contrast to temperate systems with distinct annual patterns, our system responds to episodic events. The 4-months observation period is representative for the whole year and covers the hydrological extremes. The observation period allows us to study the effect of meteorological conditions of 2009 till 2018 on N cycling using the calculated N_{surpl} of 2017. The calibration process and extrapolation onto the study area are explained in detail in section 3.3.

3.2. Model Input Data

All data sets and parameters are listed in **Table 1**. Data were lumped for Bucca Bucca and study site sub-catchments using GRASS GIS (GRASS Development Team, 2017) performing GIS operations. Temporary variable input data was downloaded for 9 years with a daily resolution 2009-01-01 to 2017-12-31. The ASTER GDEM (Tachikawa et al., 2011) with a 30×30 m resolution was used to delineate the main eastern flowing river catchments of NSW within the GDR. Within the study area, we used a 5×5 m DEM (Geoscience Australia, 2015) to calculate 217 sub-catchments. AWRP precipitation and temperature data have an original resolution of 5×5 km, which does not match with the size of the sub-catchments ranging from 0.56 to 8.56 km^2 . Hence, the resolution was refined to 5 m and smoothed out using the nearest-neighbor interpolation. Local weather anomalies were neglected. Daily minimum and maximum air temperature were used to calculate daily soil temperature in the middle of each soil layer (15 and 65 cm) (Horton and Corkrey, 2011). Evapotranspiration was available from the MODIS product MOD16A2 (Running et al., 2017) in an 8-days resolution. Evaporation was assumed to be constant for 8 days

TABLE 1 | Abbreviations (Abbr.), unit, grid size, data ranges from 0.3 m topsoil (TS) and 0.7 m subsoil (SS) and sources from soil, meteorological and conceptualization data used for the model.

Data	Abbr.	Unit	Grid Size	Range TP	Range SS	Source
SOIL PROPERTIES						
Cation exchange capacity	CEC	meq/100 g	90 × 90 m	4.80–10.85	5.04–11.99	Rossel et al., 2015
Soil organic content	SOC	%	90 × 90 m	1.79–4.43	0.56–0.98	Odgers et al., 2015
Sand	SND	%	90 × 90 m	36.92–80.61	29.76–77.20	Rossel et al., 2015
Silt	SLT	%	90 × 90 m	7.68–30.97	6.69–29.02	Rossel et al., 2015
Clay	CLY	%	90 × 90 m	10.77–33.98	16.61–42.31	Rossel et al., 2015
Bulk density	BDW	g/cm ³	90 × 90 m	1.05–1.30	1.25–1.42	Rossel et al., 2015
pH–CaCl ₂	PHC	–	90 × 90 m	4.14–5.37	4.20–5.57	Rossel et al., 2015
Organic matter	OM	%	90 × 90 m	3.08–7.63	0.97–1.68	SOC * 1.72 (Walkley and Black, 1934)
pH	PH	–	90 × 90 m	4.94–6.17	5.00–6.37	PHC + 0.8 (Laffan, 2016)
Residual water content	WC _{res}	cm ³ /cm ³	90 × 90 m	0.09–0.18	0.11–0.22	0.22733 – 0.00164 * SND + 0.00235 * ECE – 0.00831 * PH + 0.000018 * CLY * SND * CLY (Hodnett and Tomasella, 2002)
Saturated water content	WC _{sat}	cm ³ /cm ³	90 × 90 m	0.45–0.53	0.41–0.48	0.81799 + 0.00099 * CLY 0.3142 – * BDW + 0.00018 * ECE + 0.00451 * PH – 0.000005 * SND * CLY (Hodnett and Tomasella, 2002)
Field capacity water content	WC _{fc}	cm ³ /cm ³	90 × 90 m	0.19–0.35	0.12–0.17	0.0738 + 0.0016 * SLT + 0.0003 * CLY + 0.03 * OM (Pidgeon, 1972)
Van Genuchten Parameter n	n	–	90 × 90 m	1.35–1.56	1.38–1.56	exp(0.62986 – 0.00833 * CLY – 0.00529 * SOC + 0.00593 * PH + 0.00007 * CLY2 – 0.00014 * SND * SLT) (Hodnett and Tomasella, 2002)
Saturated hydraulic conductivity	K _{sat}	cm/d	90 × 90 m	13.98–51.81	3.57–15.12	10^(0.40220 + 0.26122 * PH + 0.44565 * TS*) – 0.02329 * CLY – 0.01265 * SLT – 0.01038 * ECE) (de Boer, 2016)
Freundlich Parameter kf	FRL _{kf}	m ³ /kg	90 × 90 m	36.04–60.87	42.61–69.96	30.841 – 1.263 * CEC + 1.235 * CLY (Vogeler et al., 2011)
Freundlich Parameter nf	FRL _{nf}	–	90 × 90 m	0.38–0.48	0.34–0.45	0.460 + 0.013 * CEC – 0.006 * CLY (Vogeler et al., 2011)
METEOROLOGICAL PARAMETERS						
Precipitation	P	mm/d	5 × 5 km**)	0–343	–	Jones et al., 2009
Evapotranspiration	ET	mm/d	30 × 30 m	0.47–8.68	–	Running et al., 2017
Temperature	T	°C	5 × 5 km**)	10.9–33.5	13.3–30.9	Jones et al., 2009
CONCEPTUALIZATION						
Land use	–	–	5 × 5 m	–	–	OEH, 2017
Digital Elevation Model	DEM	m	5 × 5 m	–	–	Geoscience Australia, 2015

*1 if topsoil layer, 0 if subsoil layer.

**Grid was refined and smoothed out to a resolution of 5 × 5 m.

(Hajati et al., 2019). Detailed land use maps are available for the year 2013 (OEH, 2017). All land was classified as (1) horticulture (agricultural and horticultural land use), (2) cleared land (all cleared land types from pasture to fertilized grazing), (3) pristine (all natural areas from nature conservation to rehabilitation), or (4) urban (all urban areas, including infrastructure).

3.3. Nitrogen Transformations

Nitrogen fertilization periods and amounts are unknown, though, from what we could determine following conversations

with farmers, the most common forms of N fertilizer in our study area are urea and ammonium based. Hence, we assume that N is applied with a ratio of 50% urea and 50% ammonium (White, 2018) at a constant daily rate. Furthermore, we assume top and subsoil are fully mixed. Hence, the changing concentration c (mg L⁻¹) over time t (day) in the top and subsoil layer neglects diffusion, advection, and attenuation. Adsorption (retardation) is considered for ammonium. N transformations are the same for top and subsoil and are introduced in the following modeling order.

- (1) Hydrolysis: Urease is the key enzyme responsible for the transformation from urea to ammonium (hydrolysis). Hydrolysis depends on soil temperature, soil water content and available urea concentration. Here, hydrolysis rate, HYDR (1 d^{-1}) was modeled combining the effect of temperature, moisture, and urea application (Lei et al., 2017):

$$HYDR = 0.157 * e^{2.251 \cdot 10^{-3} \cdot |834.033 \cdot \theta| + 2.885 \cdot 10^{-3} \cdot |422.949 - UREA| - 0.041 \cdot |51.016 \cdot T|} \quad (1)$$

with θ the soil water content (-), UREA the urea concentration ($\text{mg L}^{-1} \text{ d}^{-1}$), and T the soil temperature of the middle of the modeled layer (K).

- (2) Volatilization: Volatilization reduces N from the aqueous phase by transforming ammonium to ammonia. A function depending on the pK_a value for ammonia ($pK_a = 0.0901821 + 2729.92/T$), which describes the ionic equilibrium between ammonia and ammonium, is used to calculate the volatilization rate VOL (1 d^{-1}). This function depends on soil temperature (K) in the middle of the modeled layer and soil pH (-) (Emerson et al., 1975):

$$VOL = \frac{1}{10^{0.0901821 + \frac{2729.92}{T} - pH + 1}} \quad (2)$$

Soil-water-ammonia will equilibrate with air-ammonia. It is assumed that all aqueous ammonia will immediately leave the water phase into the atmosphere, because atmospheric ammonia concentrations are low. Hence, volatilized ammonium is considered a quick loss into the atmosphere.

- (3) Ammonium retardation (adsorption and desorption): Retardation is only relevant for ammonium and is driven by adsorption and desorption of ammonium on soil particles. The retardation amount depends on soil pH (-), clay content (-), and organic carbon content (-) (Sheppard et al., 2009). A pedo-transfer function was used to calculate the Freundlich parameters k_f ($\text{m}^3 \text{ kg}^{-1}$) and n_f (-), which takes the cation exchange capacity ($\text{meq}/100 \text{ g}$) and the clay content (-) into account (Vogeler et al., 2011). The Freundlich parameters were used to calculate the amount of nitrogen that is adsorbed on soil:

$$SORP = k_f * c_{NH_4}^{n_f} * BD * V_{tot} \quad (3)$$

with SORP (mg-N m^{-2}), BD the bulk density of top and subsoil (kg/m^3) and V_{tot} the total volume of the top and subsoil with 300 and 700 mm, respectively. Because of the nature of the Freundlich equation, the equilibrium concentrations change greatly at low ammonium concentrations. To prevent numerical instability, we limited the daily ad- and desorption load, which was initialized by an aqueous ammonium concentration change, to 10%. This limit delays the ad- and desorption process,

keeping nitrogen dissolved in water and bound to soil for a longer time.

- (4) Nitrification: Nitrification is the transformation from ammonium to nitrate by autotrophic archaeal (AOA) and bacterial ammonia-oxidizers (AOB). Nitrifying bacterial ammonia-oxidizers are most efficient if soil water content equals field capacity with no activity at air-dry soil and a 10% activity at saturation (Riedo et al., 1998). Temperature also has a positive effect on nitrification and follows the Van't Hoff-Arrhenius relationship (Keenan et al., 1979). An increase in ammonium concentration has a linear effect on the increase of nitrification activity. Furthermore, the efficiency of nitrifying bacteria strongly decreases with decreasing soil pH (Meyerhof, 1916; Engel and Alexander, 1958; Wild et al., 1971). Recent research showed that AOA are found in acid soils, which, with an absence of AOB, are the reason for high nitrification rates even in seasonally wet and acid tropical soils (Nicol et al., 2008; Gubry-Rangin et al., 2010; Yao et al., 2011; Prosser and Nicol, 2012; Pajares and Bohannan, 2016). In lights of this evidence by recent research and high NO_x creek concentrations, we neglected nitrification reduction by a decreasing pH. Hence, the nitrification rate NITRI (mg-N d^{-1}) is a function of soil water content ($f_{nitri,\theta}$), soil temperature ($f_{nitri,T}$) and availability of ammonium (NH_4 [mg-N]):

$$NITRI = f_{nitri,\theta} * f_{nitri,T} * NH_4 \quad (4)$$

- (5) Denitrification: Anaerobic bacteria transfer nitrate to molecular N (denitrification) and eliminate N from the aqueous phase. The bacterial activity depends primarily on temperature ($f_{deni,T}$), soil pH ($f_{deni,pH}$), oxygen ($f_{deni,\theta}$), carbon (k_{deni}), and nitrate (NO_3 [mg-N]) availability:

$$DENI = k_{deni} * f_{deni,\theta} * f_{deni,T} * f_{deni,pH} * NO_3 \quad (5)$$

Denitrification is a sub-oxic process, and it is assumed to be negligible if the pore space is filled with air (water content is below field capacity). If the water content is within field capacity and saturation, the denitrification rate DENI (mg-N d^{-1}) increases linearly (Riedo et al., 1998). The effect of soil pH on denitrification is described by a bell-shaped function. Maximum denitrification ratio occurs at $\text{pH} = 7$, a reduction of 50% occurs at soil $\text{pH} > 8$ or < 6 and a very low activity occurs at a soil $\text{pH} < 5$ and > 9 (Peterjohn, 1991). The denitrification activity increases linearly with increased available nitrate, but it is observed that at very high NO_3 concentrations (e.g., $100\text{--}500 \text{ mg L}^{-1} \text{ d}^{-1}$ Bernot and Dodds, 2005) denitrification saturates and no longer linearly increases. It is recommended to use the Michaelis-Menten form to describe the effect of NO_3 availability on denitrification (Heinen, 2006). Our catchments are assumed to be below very high concentrations, even in heavily fertilized horticulture catchments. Our estimations may therefore overestimate denitrification if very high concentrations were reached. The effect of temperature on denitrification is described by the Van't Hoff-Arrhenius

relationship (Dawson and Murphy, 1972). The model does not include the carbon cycle. However, the model parameter k_{deni} accounts for soil carbon content, derived in a calibration process.

3.4. Observation Catchments, Model Calibration, and Extrapolation

To calibrate the model, measured NO_x creek water concentrations were fitted with modeled NO_3 leakage to creeks. Observation catchments are nine pristine and seven agriculturally-dominated catchments located in the Bucca Bucca region (see **Figures 1B,C**, showing creeks A–G), which are described in White et al. (2018). These 16 catchments have weekly creek NO_x measurements available for 4 months from 2017-02-07 until 2017-05-07, with a wide range of land uses and a similar geomorphology and climate to the study site. Within the 4-months measurement period, the air temperature was on average 23.3°C (standard deviation 3.4°C), which is slightly above the annual average temperature of 21.5°C (standard deviation 4.8°C).

The model generally applied physical and empirical equations calculating N transformations in soils. The only unknown variables are N input and the effect of soil organic carbon (SOC) on denitrification. The calibration finds best values for catchment N_{surpl} and k_{deni} , which are directly related to N input and SOC on denitrification, respectively. Hence, the calibration results in the best N_{surpl} and k_{deni} values for each Bucca Bucca catchment, with which the closest match of measured daily NO_x and modeled NO_3 creek water concentrations are simulated. The model code was interpreted as validated if fitting the two unknown values N_{surpl} and k_{deni} , i.e., (1) reproduces N leakage events to creeks by the same observed precipitation events and (2) creek NO_x concentrations rising and falling at the same order of magnitude. We fitted measured NO_x with modeled NO_3 creek water concentrations rather than NH_4 creek water concentrations. Regional creek water NO_3 is assumed to be modified by excess N inputs into the catchment, while NH_4 can be a result of internal processes (White et al., 2018; Wadnerkar et al., 2019). The model performance of the calibration was evaluated using the Nash–Sutcliffe model efficiency coefficient (NSE) (Nash and Sutcliffe, 1970):

$$NSE = 1 - \frac{\sum_{t=1}^n (M_i * O_i)^2}{\sum_{t=1}^n (O_i * \bar{O})^2} \quad (6)$$

With n the number of measurements, M_i the modeled value at day t , O_i the measured value at day t and \bar{O} the mean of all measured values. If $NSE = 1$, the model simulates observation values perfectly, if $NSE = 0$, the model has the same explanatory power than the mean of all observations and if $NSE < 0$ the model performance is a worse predictor than the observation mean. Hence, the closer NSE is to 1 the better the model performance.

To calibrate the models of the Bucca Bucca catchments, we ran the model 500 times with different random value combination for N_{surpl} and k_{deni} . These random values were chosen within a reasonable data range based on the land use types pristine (all kinds of forests) and non-pristine (horticulture and grazing).

Since fertilization practices on cleared land for grazing or on blueberry horticulture is unknown, we assume a constant daily fertilizer application rate for non-pristine land use: k_{deni} values were chosen to range between 0.001 and 20, a common data range (Heinen, 2006). The random N_{surpl} values ranged according to the lumped land use from $2.25 \text{ kg-N ha}^{-1} \text{ a}^{-1}$ for entirely pristine to $121 \text{ kg-N ha}^{-1} \text{ a}^{-1}$ [recommended fertilization rate for Australian blueberries (Doughty et al., 1988; Ireland and Wilk, 2006)] for entirely agricultural land use. Pristine Bucca Bucca creeks respond to rain events with NO_x leakage, implying an open N cycle (White et al., 2018). Hence, we assume a daily input rate of $2.25 \text{ kg-N ha}^{-1} \text{ a}^{-1}$ for pristine catchments. Random values of k_{deni} and N_{surpl} within the defined value ranges guarantee no bias in the interpretation.

The difference between modeled NO_3 and measured NO_x was calculated for all 500 model runs at the measured dates. The model runs resulted in one optimum value of N_{surpl} and k_{deni} for each Bucca Bucca catchment that simulates the closest NO_x concentration. The optimum values visualized the effect land use has on the catchments. This correlation of land use with the parameters N_{surpl} and k_{deni} at the 16 Bucca Bucca catchments were up-scaled to the 217 sub-catchments within the study area. The up-scaling is reasonable because the observation and modeled catchments have similar historical land use changes (i.e., heavily-fertilized blueberry horticulture), the same climate, similar stream orders, and the same geomorphology (White et al., 2018; Conrad et al., 2019).

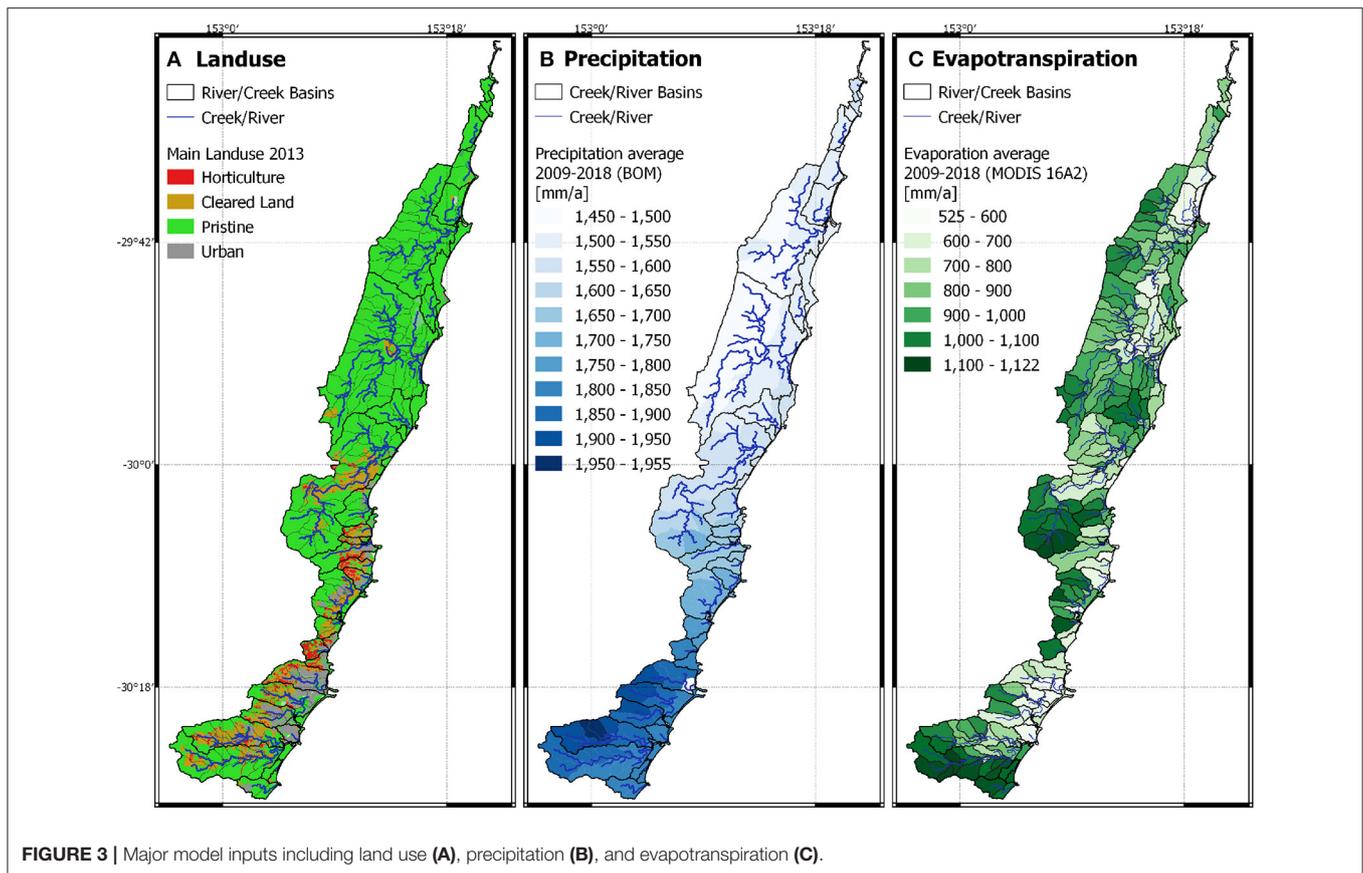
4. RESULTS

4.1. Model Inputs and Model Performance

The spatial data reveals that the study site is characterized in the North by forest with less precipitation and higher evapotranspiration than the South. The South has more urban and grazing areas than the North (**Figure 3**). Annual precipitation ranged during the 9-years modeled period between 1450 and 1955 mm, and evapotranspiration ranged from 525 to 1122 mm. The daily averaged soil temperature at 15 cm increased from South to North from 19.9 to 21.6°C and at 65 cm from 20.3 to 21.8°C , respectively. Topsoil pH ranged from 4.9 to 5.8, and subsoil pH ranged from 5.0 to 6.0. The highest pH values occurred at the southern mountain range and along the coastline, while lowest pH values occurred in the northern forests.

The study site is covered by 77% forest, 12% cleared land, 3% horticulture, and 8% urban areas. The current change in land use toward intensive horticulture has increased N losses to open waters and the atmosphere. Pristine forests in the North lose $2.25 \text{ kg-N ha}^{-1}$ on average, while horticulture loses 92 kg-N ha^{-1} . If the entire study site remained pristine, the study site would emit annually 42,006 and 120,338 kg-N to waters and atmosphere, respectively. With the current land use the study site emits 147,022 and 805,581 kg-N to water and atmosphere, respectively. Thus, N release to creeks and atmosphere increases 3.5- and 6.7-fold, respectively, originating from 3% horticulture land use.

Of all precipitation events during the observation period, only one exceeded >100 mm. This major event increased nitrate



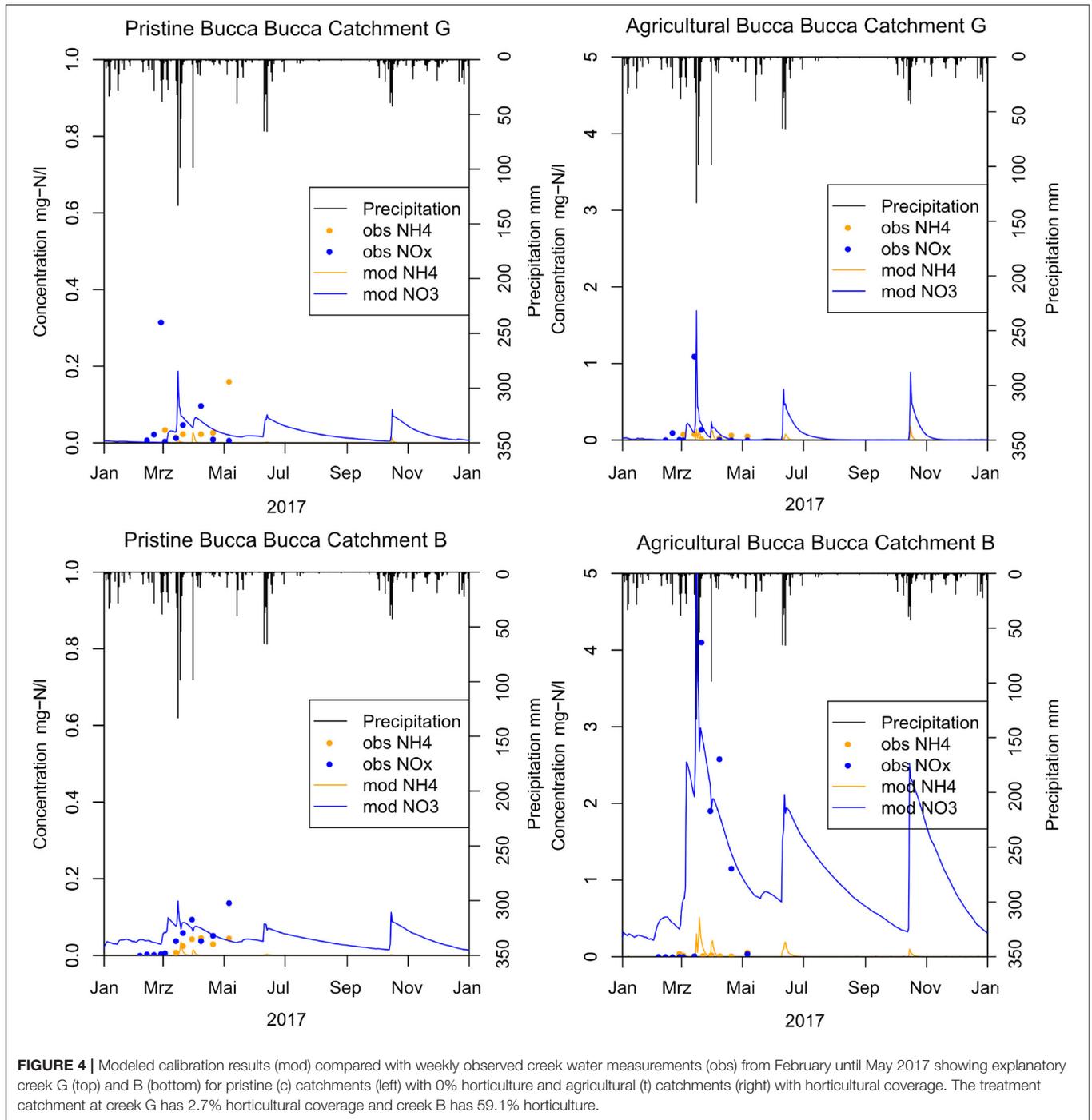
loads and concentrations in all 16 Bucca Bucca creeks. This precipitation NO_3 response was reproduced in the model for pristine and agricultural catchments (Figure 4). The modeled creek NO_3 concentration followed the trend of observations in all 16 Bucca Bucca catchments, even though the tail of decreasing concentration was slower in the model than observed. The mean NSE for eight agricultural and nine pristine catchments is 0.18, while the mean for agricultural and pristine catchments is 0.55 (standard deviation = 0.31) and -0.15 (standard deviation = 0.61), respectively. The model performance increases with agricultural coverage and has very good performance for fertilized catchments and very low performance in pristine catchments.

4.2. Correlation of Land Use With Nitrogen Surplus and Denitrification Potential

Fitting NO_x measurements of the Bucca Bucca creeks with the modeled creek nitrate concentrations revealed optimum values for catchment N_{surpl} and the k_{deni} . As expected, catchments with less horticulture coverage showed lower N_{surpl} (Figure 5). Catchments with larger horticultural coverage (creeks B, C, D, and E) had higher NO_x flushing after precipitation events and showed continuously larger modeled NO_x concentrations at dry periods (Figure 4, bottom right), on average 0.09 mg-N L^{-1} . Optimum annual catchment N_{surpl} ranged from a minimum of $0.12 \text{ kg-N ha}^{-1} \text{ a}^{-1}$ in a pristine catchment (creek F_C) to a

maximum of $66.02 \text{ kg-N ha}^{-1} \text{ a}^{-1}$ in an agricultural catchment (creek $D_{T,2}$). The model coefficient k_{deni} , ranged from 0.17 to 10.91, with a mean value of 4.40. The regression analysis of N_{surpl} with horticulture, cleared land and forest resulted in p -values of $p < 0.001$, $p = 0.95$, and $p = 0.003$, respectively. Since N_{surpl} has the most significant correlation with horticulture area (Figure 5, bottom), we extrapolated the linear trend onto the study site. The annual N_{surpl} of catchments without horticulture ranged between 0.12 and $4.38 \text{ kg-N ha}^{-1}$, with a mean of $2.25 \text{ kg-N ha}^{-1} \text{ a}^{-1}$ ($0.6 \text{ mg-N ha}^{-1} \text{ d}^{-1}$). This mean value was extrapolated onto catchments without horticulture across the study site.

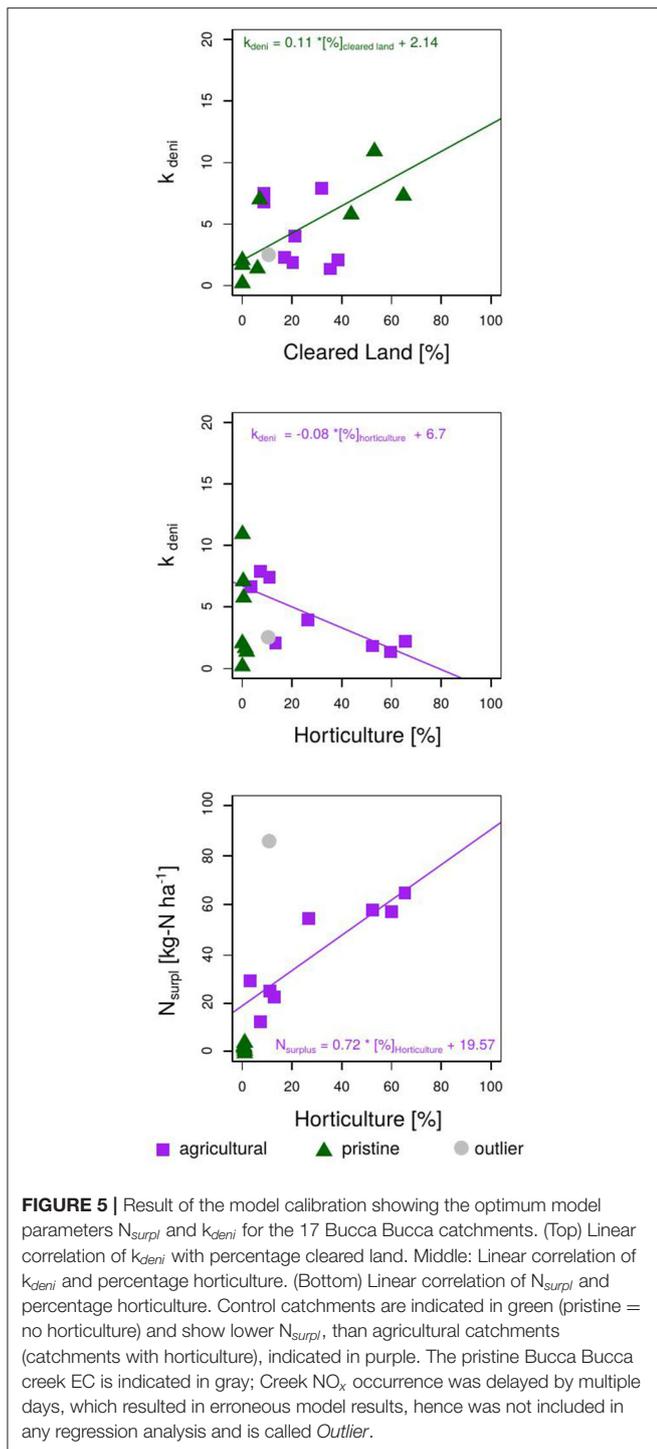
The regression analysis of k_{deni} with horticulture, cleared land, and forest resulted in p -values of 0.16, 0.07, and 0.89, respectively, indicating only a correlation of low significance between k_{deni} and percentage cleared land. Excluding the pristine catchments from the regression analysis of k_{deni} and percentage horticulture, the significance increases ($p = 0.02$). Excluding the agricultural catchments from the correlation between k_{deni} and coverage cleared land, the significance increases also (p -value = 0.03). This indicates that the land uses, horticulture and cleared land, show different effects on the denitrification potential (k_{deni}). Non-agricultural sites with forest coverage have low k_{deni} values (0.17–2.05), and non-agricultural sites of mainly cleared land have high k_{deni} values (5.78–10.92). A high k_{deni} implies a higher reduction of N through denitrification, which



is highest at sites with no horticulture and high coverage of cleared land. There is thus a negative and positive correlation between k_{deni} and increasing horticultural and cleared land coverage, respectively. We chose to extrapolate the significant correlation of k_{deni} and coverage cleared land at sites without horticulture (Figure 5, top) and the significant correlation between k_{deni} and horticulture on catchments with horticulture (Figure 5, middle).

4.3. Model Extrapolation to Regional Catchments

We applied the derived linear correlations from the Bucca Bucca catchments to calculate k_{deni} for catchments without horticulture as a function of cleared land coverage, and for catchments with horticulture as a function of horticulture coverage. N_{surpl} was calculated as a function of horticulture coverage. A total of 71 of the 217 sub-catchments are partly covered with horticulture,



hence have higher N_{surpl} than areas with only forest and cleared land (Figure 5, bottom). NH_4 and NO_3 leaching into creeks and rivers is high in the South and low in the North (Figure 6). The lowest N_{surpl} ($19.6 \text{ kg-N ha}^{-1} \text{ a}^{-1}$) was calculated for a catchment with 0.07% horticulture, and the highest N_{surpl} ($53.1 \text{ kg-N ha}^{-1} \text{ a}^{-1}$) was calculated for a catchment with 46.7% horticulture.

The 900 km^2 study area has an estimated N_{surpl} of $980,000 \text{ kg-N a}^{-1}$, 87% of which originates from catchments with horticulture, even though horticulture covers only 3% of the total area. $149,000 \text{ kg-N a}^{-1}$ are lost into surface water bodies. Losses into the atmosphere via volatilization and denitrification are 5-fold higher than surface water, at $806,000 \text{ kg-N a}^{-1}$, where 91% originates from catchments with horticulture. Hence, catchments with horticulture release less N to creeks than into the atmosphere, but far more than pristine catchments (Figure 7).

The water balance of the model CoCa-RFSGD calculates for catchments with horticulture more surface runoff than interflow (soil drainage), 55.6 and 44.4%, respectively. In contrast, pristine catchment creek discharge originated 40.6% from immediate runoff and 59.4% from interflow. The form of N discharged into creek water also differed between different land use types. Catchments with horticulture and catchments with only cleared land and forest showed relative losses of 83 or 96% nitrate, 15 or 4% ammonium, and 1.2 or 0.07% urea, respectively.

Soil properties and meteorological conditions affected N transformations. Relative loads into air increased by increasing soil pH, soil depth, precipitation surplus, and residual water content (Equations 1–5). Hence, N transformations differed regarding land use, soil type, and meteorological conditions (see Table 2). Nitrification and denitrification were higher during the wet season in horticultural catchments, while volatilization and hydrolysis did not change. Volatilization was only 0.02% of the annual N_{surpl} . Non-horticultural catchments retarded about 38% of the modeled N input during the wet and dry season. Horticultural topsoils show adsorption and desorption processes at equilibrium (Table 2).

The effect of wet and dry seasons on N leaching into surface water bodies can be interpreted with the model. Catchments with horticulture released during the wet season from December to May 66 and 67% ammonium through surface runoff and interflow, respectively. Pristine catchments released during the wet season 55 and 50% ammonium through direct runoff and interflow, respectively. Catchments with horticulture released nitrate during the wet season 53 and 50% through direct runoff and interflow, respectively, while catchments without horticulture released 44 and 55% nitrate through direct runoff and interflow, respectively.

Monthly estimates revealed for the study year 2017, the highest releases of N through direct surface runoff and interflow were in March, June, and October, the wettest months of that year (Figure 8). Highest loads to creeks were in March, which was part of the measurement period from February to May. Lowest N losses to atmosphere were in the winter months with lower temperatures. Catchment N loads from horticultural catchments to creeks and rivers vary largely over the model years from 2009 till 2017 (see Figure 9). The wettest year, 2009 (on average $2,209 \text{ mm}$), and the driest year, 2016 (on average 1082 mm), did not show the largest and lowest N loads to surface water bodies. The highest loads for surface water bodies were in 2015 with 5.4 kg-N ha^{-1} and the lowest in 2013 with 1.9 kg-N ha^{-1} , and the highest loads to atmosphere occurred in 2016 with $24.2 \text{ kg-N ha}^{-1}$ and the lowest loads in 2015 with $12.8 \text{ kg-N ha}^{-1}$.

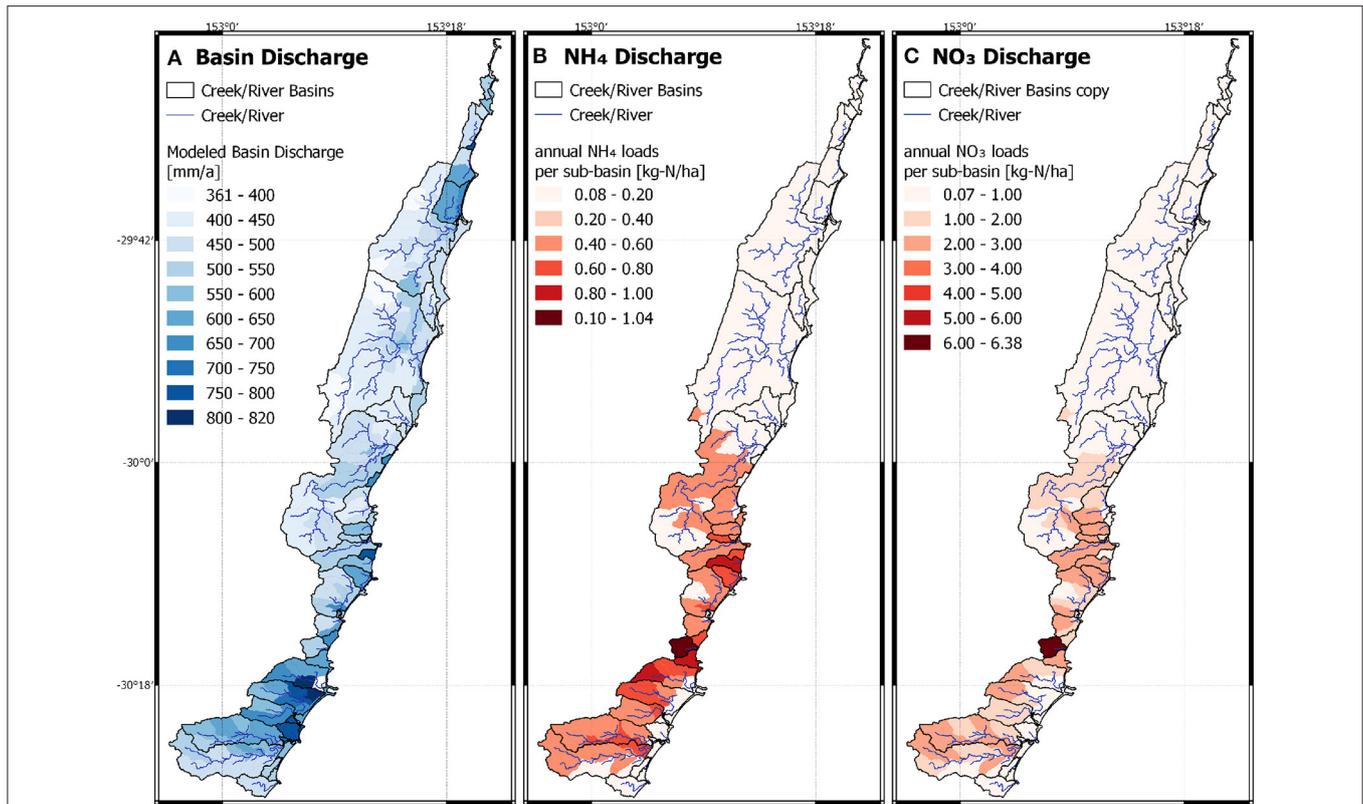


FIGURE 6 | Model result of the extrapolation of the calibrated Bucca Bucca catchments applied on the study area. **(A)** Averaged annual discharge into creeks by immediate surface runoff and interflow. **(B)** Averaged annual discharge of ammonium into creeks. **(C)** Averaged annual nitrate discharge into creeks.

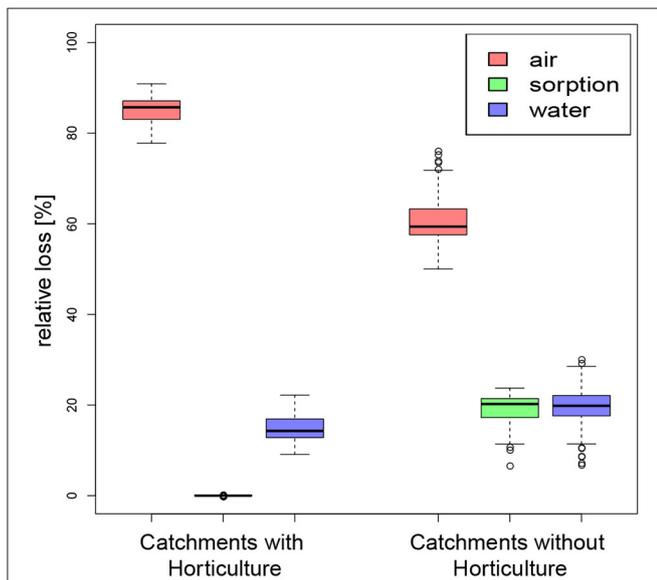


FIGURE 7 | Relative losses of nitrogen into air, water, or retardation from catchments with and without horticulture, indicating a larger importance of volatilization and denitrification (losses into air) for catchments with horticulture than for pristine catchments.

5. DISCUSSION

Nitrogen losses to creeks across a land use gradient in subtropical catchments were analyzed by upscaling the correlation of land use with water quality measurements of low-order creek catchments. The large N loads imply an open N cycle even for pristine catchments but increasingly with increased coverage of horticulture. Hence, an over-application of N fertilizers is likely the reason for NO_x in surface water bodies. The model built in this study takes into account daily soil moisture changes, meteorological values and soil properties to calculate N transformations and transport. This model can be extrapolated to areas with a similar land use and a similar climate.

5.1. Importance of Subtropical Climate and Soil

Soil pH, temperature and moisture are important factors impacting microbial processes and N transformations in the model, and their effect has also been observed in *in situ* studies. For example, soil pH impacts volatilization. Measurements at an acid soil in Canada resulted in volatilization losses between 3 and 7% (Fan and Mackenzie, 1993). The soil of the study area has low pH typical of tropical soil lowering N losses by volatilization to 0.02% and rendering this transformation locally irrelevant.

TABLE 2 | Percentage of annual nitrogen surplus (N_{surpl}) being transformed by hydrolysis, volatilization, nitrification, denitrification and retarded by ad- and desorption processes.

		Hydrolysis		Volatilization		Retardation***		Nitrification		Denitrification	
		Wet	Dry	Wet	Dry	Wet	Dry	Wet**	Dry	Wet	Dry
Non-pristine	TS	49.5	49.7	0.0	0.0	0.0	0.28	100.0	80.8	69.0	41.7
	SS	0.2	0.3	0.0	0.0	0.0	-0.1	10.1	4.2	34.5	25.5
Pristine*	TS	49.6	49.8	0.0	0.0	38.6	38.0	81.5	73.9	34.7	21.7
	SS	0.2	0.2	0.0	0.0	0.2	0.1	2.8	1.4	37.5	27.6
All	TS	49.5	49.7	0.0	0.0	26.0	25.7	87.6	76.2	45.9	28.3
	SS	0.2	0.3	0.0	0.0	0.1	0.1	5.2	2.3	36.5	26.9

Results are split into wet (Dec.–May) and dry (Jun.–Nov.) season and into topsoil (TS) and subsoil (SS).

*No horticultural land use.

**Nitrification can be higher than 100%, because of desorbed N from the dry season.

***Retardation is negative if there is more N desorbed annually, positive if there is more N adsorbed annually, and zero if desorption equals adsorption.

Denitrification is also reduced at low soil pH, which has been observed in laboratory experiments (Remde and Conrad, 1991). The topsoil of the study site has a pH ranging from 5 to 6.2, potentially inhibiting nitrification (Meyerhof, 1916; Engel and Alexander, 1958; Wild et al., 1971). However, recent research showed high nitrification rates even in strongly acid soils due to autotrophic bacteria (Pett-Ridge et al., 2013), or more likely, specialized archaea (Gubry-Rangin et al., 2010; He et al., 2012; Hu et al., 2014; Stempfhuber et al., 2015; Li et al., 2018). Taking into account low NH_4 and elevated NO_x creek measurements after a heavy rain event, we assume nitrification is not limited by pH at our study site. Soil temperatures and soil moisture also impact volatilization, nitrification, and denitrification by increasing microbial activity (Talling et al., 1998; Rotz, 2004; Montes et al., 2009, e.g.) and denitrification by changing oxygen availability for the microbial community (Jørgensen et al., 1998; Allen et al., 2010; Hansen et al., 2014), respectively.

Modeling results reveal that nitrate discharge into creeks did not change in wet and dry season, while 2/3 of the ammonium discharged to creeks during the wet season. Denitrification increases with higher nitrate concentrations, which reduces the potential for nitrate accumulation during the dry season and release during the wet season. During the wet season, ammonium pore water concentrations are diluted, and the soil can adsorb less ammonium resulting in lagged desorption of soil ammonium. This desorbed ammonium has the potential to be nitrified, delaying pollution loads to creeks until the wet season arrives and transports N from soil storages to the creeks. During months with high precipitation, the highest loads of N are transported into creeks (Figure 8). Seasonal N loads to rivers have been observed to affect N transformations in unregulated and small rivers elsewhere. For example, 85% of the annual river and most of the nitrate discharge was observed during the wet season from November to April in southwest Europe (Romero et al., 2013). In contrast, highly regulated rivers retain most of the nitrate in their catchment and have relatively low nitrogen concentrations (Lassaletta et al., 2009).

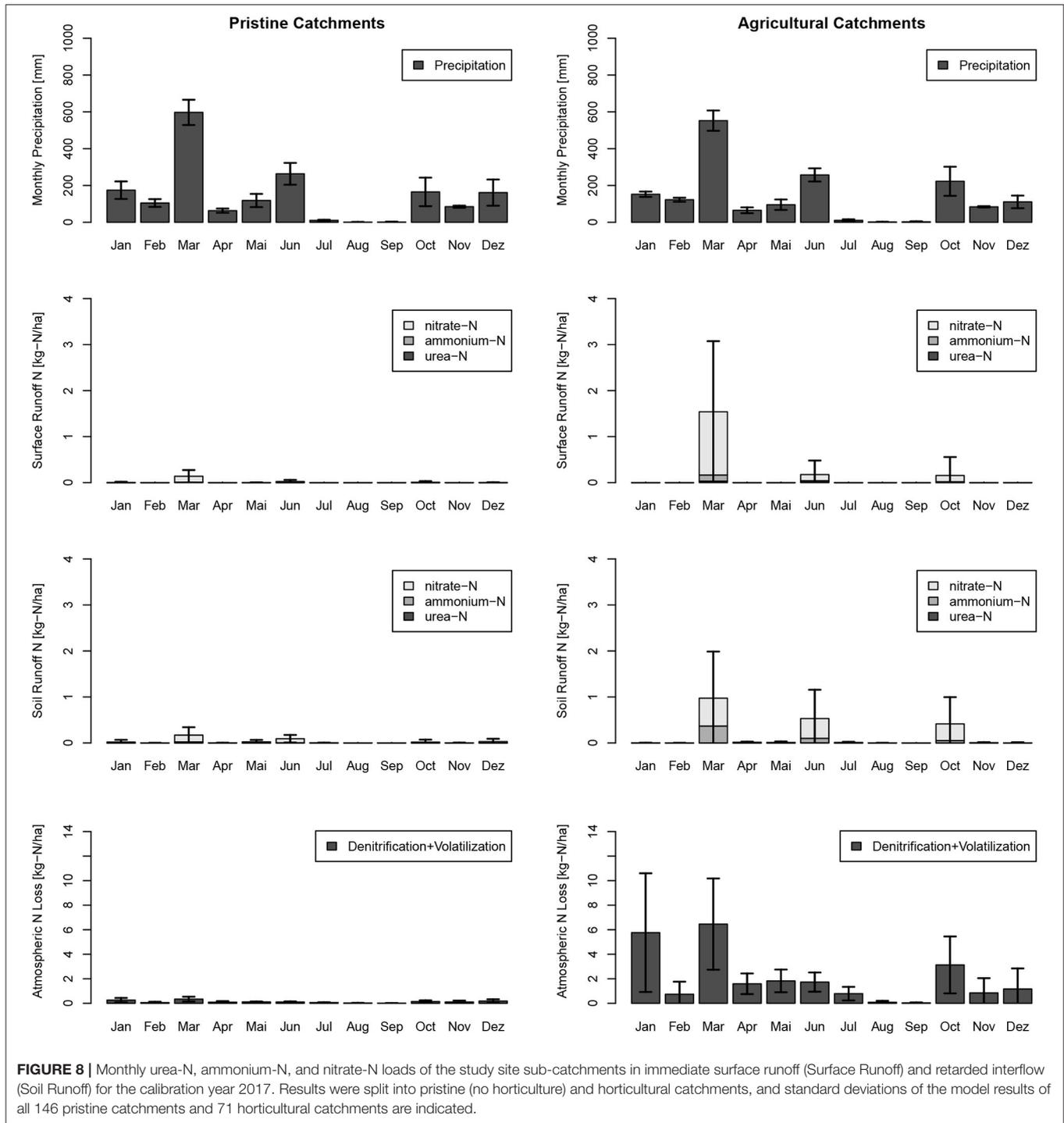
Prolonged droughts and increased extreme river runoff for many regions of the world are predicted for the next decades

(Davidson et al., 2011), which may lead to increased nitrogen build-up in soils during the dry season and an increased sudden release of nitrate and ammonium during flooding events. The additional reactive N from horticulture will mainly transfer to the atmosphere rather than build up in soil due to higher denitrification potential (Figure 7), which would proportionally reduce stream loads (Lee et al., 2019). In contrast, desorption and nitrification of ammonium during the wet season will increase surface water nitrate concentrations. A large fraction of the nitrate losses into surface water bodies are expected to be denitrified at higher temperature and nitrate concentrations, which makes it an important process in tropical waters (Talling et al., 1998). Hence, the creek nitrogen loads in the study site may be reduced in a warming climate, replaced by atmospheric losses. Depending on the soil and sediment conditions, this reactive N may be exported as N_2 or the greenhouse gas N_2O .

Three pristine Bucca Bucca catchments showed 4.6, 137.6, and 19.6 times higher NO_x concentrations in creek water during the wetter months (February and March) compared to the dryer months (April and May) during observations (White et al., 2018). These concentration gradients were incorporated in the model by assuming an annual N_{surpl} of 2.25 kg-N ha^{-1} even for entirely pristine catchments. The *Soil Landscapes of the Coffs Harbour* dataset (OEH, 2018) describes the soil of the study area to be strongly acidic, which is typical for tropical and subtropical soils. The older underlying geologic substrates are generally depleted in phosphorus and cations (Pajares and Bohannan, 2016). Hence, tropical and subtropical soils tend to be P limited with relatively high losses of N via denitrification and groundwater leaching (Vitousek and Matson, 1988; Martinelli et al., 1999; Brookshire et al., 2012). The creek water measurements in the Bucca Bucca catchments confirm that soils in pristine forests are likely N-saturated, and they function as a long-term N source, which is typical for tropical forests (Hedin et al., 2009).

5.2. Regional Importance of Intensive Horticulture

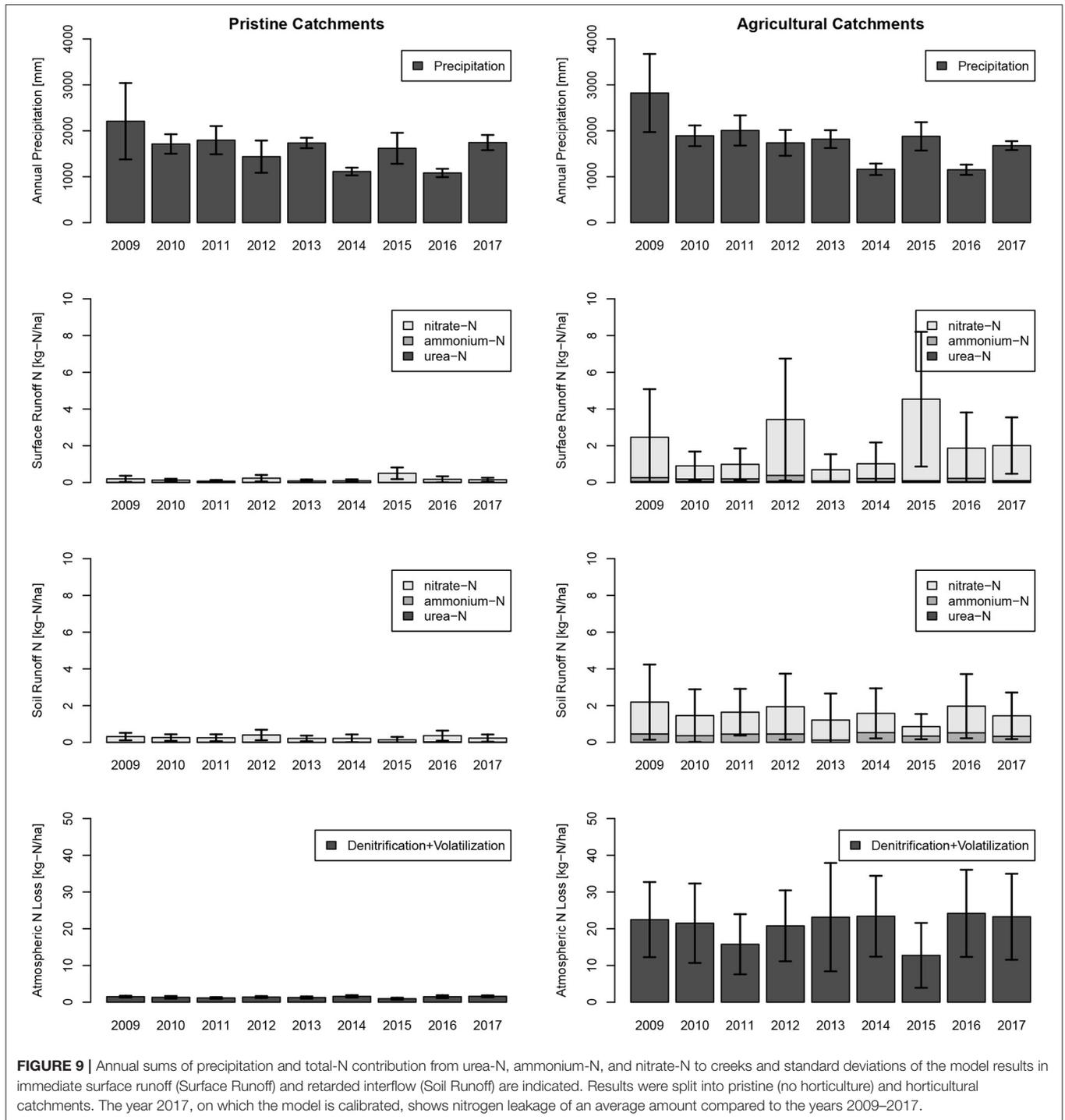
Our model implies that regional blueberry horticulture loses annually 92 kg-N ha^{-1} , which is equivalent to 76% of the



recommended application rate of 121 kg-N ha⁻¹. N_{surpl} is the total N, which was not retained within the catchment, without differentiating the source (fertilization, biological N fixation and/or atmospheric N fixation). Studies on blueberry farming showed an optimum growth:fertilizer ratio is at an annual N fertilizer rate of 63–93 kg-N ha⁻¹. Higher fertilization generates more harvest at the cost of increasing N losses

(Vargas and Bryla, 2015), potentially explaining our high nitrogen losses.

Globally, the fertilizer demand increases by 1.9% and the fertilizer supply increases by 1.7% per year (FAO, 2001). Australia follows this trend showing a growing consumption of nitrogen-based fertilizers from 47.6 kg-N ha⁻¹ in 2002 to 68.1 kg-N ha⁻¹ in 2016 (worldbank.org, 2019), representing an annual



increase of 1.5%. The increased fertilizer application trend has been observed in the regional sediment record with an 8-fold increase of phosphorous in lake sediments following blueberry farming expansion in the last two decades (Conrad et al., 2019). Nitrogen burial in sediments did not increase because nitrates are soluble and will be exported to the ocean or lost to the atmosphere (White et al., 2018). This

observation is consistent with our model results that catchments with horticulture lose 85% of the N_{surpl} into the atmosphere and pristine catchments lose 61% into the atmosphere. If horticultural areas expand by 1%, losses to the atmosphere will increase by $1.3 \text{ kg-N ha}^{-1} \text{ a}^{-1}$, while losses to surface water bodies increase by $0.2 \text{ kg-N ha}^{-1} \text{ a}^{-1}$. While most N is lost primarily to atmosphere in the tropics by increasing

denitrification, temperate climates lose more N to rivers as DIN (Lee et al., 2019).

The regional catchments have an 8% decline in denitrification potential in areas with increased horticultural land use, and an 11% increase in denitrification potential in areas with increased cleared land (Figure 5). The denitrification potential depends on soil organic carbon (SOC). SOC delivers energy to the denitrifying microbial community and thus increases denitrification. Land use changes have been reported to influence SOC, where agricultural lands have low SOC storage (Conant et al., 2017), whilst wet forest can store >25% more SOC than agricultural land (Detwiler, 1986; Penman et al., 2003; Don et al., 2011). Wet forests can store between 9% larger (Guo and Gifford, 2002) and 12% lower (Don et al., 2011) SOC than grassland, depending on local conditions. The improved denitrification potential of cleared land shows that grassland has a beneficial effect on soil organic carbon content at the study region (Conant et al., 2001).

5.3. Model Limitations and Strengths

Our model estimates N_{surpl} at a coastal area of NSW, Australia, and provides a tool to better interpret catchment responses of nitrogen leakage to creeks in poorly studied subtropical and tropical climates. The model relies on the assumption that study site streams are comprised of surface runoff and interflow. Water fluxes are calculated using the model CoCa-RFSGD (Hajati et al., 2019). These fluxes depend on storage capacity and soil properties and not on surface topography. The model presented in this study has a series of simplifications and assumptions regarding model conceptualization and nitrogen transformation modeling.

The conceptualization of the catchment hydrology was simplified by neglecting groundwater recharge and discharge to creeks. This simplification is acceptable because the modeled catchments (Figure 1) have a fast discharge response to precipitation. Groundwater recharge and discharge processes are minor as implied from tracer observations (White et al., 2018). The model calculates the soil N cycle based on a mass balance, where N transformations are calculated with common process knowledge. Catchment nitrogen surplus is calculated using creek NO_x measurements, while modeled atmospheric losses are not cross-checked with measurements. Even though it would be necessary to fit modeled atmospheric losses with measurements, this is an acceptable limitation since atmospheric losses are largely driven by temperature, soil moisture and soil pH, which are included in our model. Also, measuring the integrated atmospheric N losses is generally not possible at catchment-scale, compared to aqueous N losses. Lumping soil and meteorological variables, as well as topsoil and subsoil set-up, neglects local phenomena (e.g., local precipitation events, the temporary pH increase by urea applications or the higher soil pH and lower permeability of the upper soil). This simplification seems reasonable, because the model calibration is done with NO_x creek measurements that integrate catchment losses. An important part of the model conceptualization is the assumption that fertilizer application occurs daily at a constant rate with a ratio of 50% urea and 50% ammonium. The amount and timing

of fertilizer application are important but not known for these farming practices. Conversations with farmers indicate that small amounts of fertilizer are added often, though they are dependent on crop needs and environmental variables. This closely matches our methodology of assuming continuous small inputs over time. As low nitrate concentrations can limit denitrification (Tomasek et al., 2017), the timing of fertilizer applications can reduce nitrate leaching (Bruulsema, 2009), which depends on the growth stage of the crop; in turn, this is indirectly taken into account by looking into the losses in creek water for precisely that particular season. However, NO_x measurements during other crop growth phases are not available, and there is no information available about the fertilizer efficiency during other growth phases.

We did not take into account the N attenuation of the riparian zone, which potentially decreases nitrogen loads to creeks (Ranalli and Macalady, 2010), hence, underestimates the catchment N_{surpl} . Furthermore, we neglected the effect of nitrate saturation on denitrification, which might overestimate denitrification and the catchment N_{surpl} . Even in highly modified catchments, denitrification saturation was not observed (O'Brien et al., 2007), implying our assumption is reasonable. The carbon cycle is highly connected with the nitrogen cycle. Even though we did not model the carbon cycle, we calibrated the denitrification potential k_{deni} , which is associated with the beneficial effect of SOC. Therefore, k_{deni} is a proxy for the impact of the carbon cycle on the N cycle. The resulting significant linear correlation between denitrification potential and land use is a simplification of a complex correlation between soil organic carbon and denitrification and likely only valid in this region. Furthermore, the model is calibrated using measurements from February to May 2017. In these months, mean soil temperatures were slightly higher than in the annual average by 1.9°C, showing likely higher atmospheric losses during the observation months than the model period. Since temperature is considered in this model, the effect on calibration is negligible.

Besides the model conceptualization and simplifications on N transformations, the model calibration and model performance are impacted by the spatial and temporal input data. During the weekly measurements in 4 months, only one rain event flushed catchment N out into creek water. Multiple other rain events did not show an increase of NO_x in creek water, likely due to low soil moisture in a dry period. Hence, it was not possible to validate the model on a second response of rain on creek water quality. The lack of validation and calibration on water fluxes raises concerns for equifinality, which cannot fully be eliminated. Equifinality is largely a problem of overparametrization (Young and Beven, 1994; Beven, 1995). Since we applied basic equations of the nitrogen cycle and only fitted two parameters (k_{deni} and $N_{surplus}$), we limited the problem of equifinality. In future simulations, it would be advisable to validate the model on additional heavy rain events. Also, at the 17 observation catchments measurements were taken at 11 days within 4 months. With the low number of observations per catchment, model performance is difficult to calculate and results in all agricultural catchments to a good model performance, while 62.5% of pristine catchments show a bad model performance (NSE < 0). The model performance analysis indicates that

model performance increases with increased fertilization. One explanation might be that in pristine catchments, riverine N transformations become relatively more important, and this is not included in the model. Furthermore, the input data of the model are coarser resolved than the sub-catchments. This low resolution and missing standard deviations of input data prevents us from including the effect of microclimate on the microbial N transformations and an uncertainty analysis. Nevertheless, annual weather differences clearly show the model dependency on seasonal temperature and precipitation cycles (see **Figures 8, 9**).

Overall, it was reasonable to extrapolate the conceptualization and calibration results to multiple regional catchments because climate, lithology and land use are similar in the observed and modeled catchments. The modeled area has land use types similar to the observed Bucca Bucca catchments, which are 0–100% forest, 0–65% horticulture, and 0–68% cleared land. The model provides insights into the general nitrogen loads of pristine and horticultural catchments in the subtropics and its response to different land uses and precipitation patterns. This model could be improved by adding carbon cycling and riparian zone processes. For a thorough study site analysis, precipitation, temperature, and evapotranspiration data need to be available in an appropriate resolution, and observation data should include multiple catchment responses on precipitation events.

6. CONCLUSIONS

We developed a modeling tool calibrated on timeseries NO_x loss measurements in low-order creeks improving our understanding of the effect agricultural and pristine land use has on catchment-wide water quality in subtropical conditions. The model extends the water balance model CoCa-RFSGD, adding soil nitrogen cycling based on well-established physical and empirical equations, and has an increasing model performance with increasing agricultural coverage. We found that 81% of the reactive N_{surpl} leaves the catchments into the atmosphere and that 15% is transported to surface water bodies. The 900 km^2 large study area is only covered by 3% horticulture, which is responsible for a 3.5-fold increase in N losses to surface water bodies and a 6.7-fold increase of N losses into the atmosphere.

The calibrated model gave insight into the general catchment behavior relative to seasonal climate and land use. Ammonium accumulated in soils during the dry season and was released during the wet season, potentially increasing nitrate concentrations in surface waters. Denitrification increased during the warmer wet season and is relatively higher in horticulture catchments than in pristine ones. Aqueous N losses occur mainly during wet months. A transformation into cleared

land increased the denitrification potential due to the addition of soil organic carbon.

Our results underline that a large percentage of applied fertilizer is lost to waterways, especially in the humid subtropics and would need to be carefully regulated to prevent fueling eutrophication. In particular, we have estimated that agricultural land uses in northern NSW lose 92 kg-N ha^{-1} into the atmosphere and surface water bodies, which is equivalent to 76% of the recommended N fertilization rate. The lost N is a sizeable economic loss and potentially a health, ecological, and climate issue.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

Research idea by M-CH and IS, with contributions from NM. Measurements and study site knowledge was supplied by SW. Model conceptualization, programming, model calibration, model extrapolation, and model analysis by M-CH. Manuscript writing by M-CH with contributions from SW, NM, and IS. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/feart.2020.00347/full#supplementary-material>

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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