Ecosystem Consequences of Tree Monodominance for Nitrogen Cycling in Lowland Tropical Forest

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Abstract

Understanding how plant functional traits shape nutrient limitation and cycling on land is a major challenge in ecology. This is especially true for lowland forest ecosystems of the tropics which can be taxonomically and functionally diverse and rich in bioavailable nitrogen (N). In many tropical regions, however, diverse forests occur side-by-side with monodominant forest (one species >60% of canopy); the long-term biogeochemical consequences of tree monodominance are unclear. Particularly uncertain is whether the monodominant plant-soil system modifies nutrient balance at the ecosystem level. Here, we use chemical and stable isotope techniques to examine N cycling in old-growth Mora excelsa and diverse watershed rainforests on the island of Trinidad. Across 26 small watershed forests and 4 years, we show that Mora monodominance reduces bioavailable nitrate in the plant-soil system to exceedingly low levels which, in turn, results in small hydrologic and gaseous N losses at the watershed-level relative to adjacent N-rich diverse forests. Bioavailable N in soils and streams remained low and remarkably stable through time in Mora forests; N levels in diverse forests, on the other hand, showed high sensitivity to seasonal and inter-annual rainfall variation. Total mineral N losses from diverse forests exceeded inputs from atmospheric deposition, consistent with N saturation, while losses from Mora forests did not, suggesting N limitation. Our measures suggest that this difference cannot be explained by environmental factors but instead by low internal production and efficient retention of bioavailable N in the Mora plant-soil system. These results demonstrate ecosystem-level consequences of a tree species on the N cycle opposite to cases where trees enhance ecosystem N supply via N2 fixation and suggest that, over time, Mora monodominance may generate progressive N drawdown in the plant-soil system.

Introduction

Tropical forests harbor an exceptional diversity of tree species and occur across a wide range of soils that vary in the availability of resources required for growth [1,2]. Plant-soil nutrient interactions are critical to the productivity of tropical forests [3,4], but how these interactions depend on tree diversity and associated functional traits is not straightforward. One critical problem lies in identifying the degree to which resource competition interacts with other biotic and abiotic mechanisms [1] to generate patterns in nutrient limitation and cycling at the ecosystem level. Late-successional monodominant forests (one species ≥60% of the canopy) in the tropics are a particularly vexing case as their distribution often defies simple environmental explanation [3,6] and their persistence is, by definition, incongruous with mechanisms thought to promote high tropical tree diversity [7–11].

Conceptual models of monodominance have long proposed that in certain environments such as infrequently disturbed areas, combinations of species traits such as large seeds, deep litter, and shade tolerance can, over time, generate positive feedback in the plant-soil system that favors these traits [10]. Such plant functional traits express fundamental tradeoffs (e.g., rapid versus slow growth) in resource (e.g., light and nutrients) acquisition and investment. It is reasonable to expect that long-term dominance by a relatively restricted, but coordinated, set of traits would have consequences for resource availability at the ecosystem level. For example, most monodominant taxa are slow growing [9,10] and have slow-decomposing litter [12,13], traits associated with high plant nutrient use efficiency, low nutrient availability in soils, and slow ecosystem-level nutrient losses [3]. These traits should have consequences for the nitrogen (N) cycle in particular because rates of detrital mineralization and plant N use efficiency govern N-limited plant growth late in succession [14]. Conversely, many diverse tropical forests worldwide display high levels of bioavailable N in soils and large hydrologic and gaseous N losses [15–18], a pattern inconsistent with N-limitation at the ecosystem level [17].

A long history of ecological research has shown that individual tree species can influence local N cycling and retention in soils and in some cases affect the balance of N inputs and losses at the ecosystem level [19,20]. For example, trees capable of symbiotic N2 fixation operate to deliver new N to the ecosystem and can also increase losses of N [21,22] but these N inputs often diminish late in forest succession [15]. In contrast, traits of many tropical monodominant species would be expected to constrain internal N turnover [9,10] but the long-term consequences for ecosystem N
loses are unknown. In theory, monodominant plant traits that act to reduce soil N to levels lower than competitors require for growth should confer a competitive advantage for N [23,24]. If true, ecosystem losses of bioavailable N (e.g., nitrate) should be reduced relative to otherwise similar diverse forests, reflecting the strength of N limitation in each [14,17]. We explore this possibility by asking whether the ecosystem N cycle in remote monodominant tropical forest is qualitatively and quantitatively different from adjacent diverse forests.

We examined the chemical and isotopic properties of small watershed ecosystems dominated by a well-known monodominant species, *Mora excelsa* (Benth.; *Caesalpinioideae*), and adjacent diverse watershed rainforests in Trinidad [5]. *Mora* has several traits which may confer a competitive advantage and modify soil resources: canopy dominance, shade tolerance, low-N tissues, slow litter turnover and a dense network of surface roots [25]. *Mora* does not form symbioses with N2-fixing bacteria or ectomycorrhizal fungi, but rather its roots are heavily colonized by endophytic and arbucular mycorrhizal fungi [26]. We sampled a total of 26 small watershed forests during five spatially extensive campaigns in the Paria River basin over a four-year period. Our analyses were designed to examine if soil N properties and N losses differ between *Mora* and diverse forests and the degree to which they originate from factors internal or external to the plant-soil N cycle.

**Methods**

**Ethics Statement**

No permits or specific permissions were required for the location or activities of study. The study was conducted on public land and complied with all relevant regulations. Our field samples consisted of soil and water collections, which do not require permits, and did not involve endangered or protected species.

**Study Area**

We studied small (5–20 ha) ungauged rainforest watersheds in the remote lower Paria River Basin (~23 km²) (~10°46’25.80”N, 61°14’57.39”W) in the Northern Range of Trinidad from 2009 to 2012. The landscape contains a dense network of small tributary watersheds and harbors the largest tract of undisturbed old-growth *Mora* rainforest in Trinidad. Mean annual precipitation (MAP) is 2750 mm distributed across distinct wet (June–December) and dry (January–May) seasons [27]. Mean annual temperature (MAT) is 25°C. Soils are Ultisols developed from low-grade meta-sedimentary phylites of Cretaceous to Jurassic age [28,29]. We selected small spatially-discrete watersheds in which vegetation was distinctly dominated by either *Mora* or diverse tree assemblages. *Mora* (~90–200 meters above sea level, “MASL”) and diverse forest watersheds (~90–300 MASL) were all upland with moderate to steep slopes (5–20%) and were not subject to flooding. All forests were old-growth with no history of logging or large natural disturbance such as fire or hurricanes [3,30]. Diverse rainforests are classified as evergreen seasonal and lower montane forest and disturbance such as fire or hurricanes [5,30]. Diverse rainforests ate to steep slopes (5–20%) and were not subject to flooding. All small spatially-discrete watersheds in which vegetation was forest watersheds. In *Mora* forests, and to a lesser extent in diverse forests, thick (~2–15 cm) and densely rooted organic horizons form within the vicinity of individual trees [25]. In the wet season of 2012 we sampled this organic horizon and mineral soils (n = 3 composited cores) associated with the rooting zone (within 1 m of the trunk or buttress) of dominant canopy trees in diverse and *Mora* forests (n = 5 plots). Field-moist, root- and rock-free soils (<2 mm) were extracted (within one day) for ammonium and nitrate in 2 M KCL. Dried soils were extracted for phosphate in weak Bray’s solution. All extracts were immediately frozen, followed by colorimetric analysis (Lachat 800 series, Zellweger Analytics) at Montana State University (MSU). Soils were also analyzed for texture hydrometrically with hexametaphosphate as a dispersant, and pH using 4:1 water: soil solution. Dried mineral soils were analyzed for %C, %N, 15N, and 18O at the MBL Stable Isotope Laboratory, Woods Hole, MA.

We sampled watershed streams during the dry seasons of 2009, 2010 and 2011 and the wet season of 2011. Water samples were collected from 1st and 2nd order streams draining diverse and *Mora* watershed forests (n = 5–14) during each survey following ref 18. Though the number of watersheds sampled varied across years, a minimum of 5 watersheds of each forest type were sampled every year. Rainfall was collected during 22 rain events over 2008–2011 in an open area at 330 MASL within ~5 km from the study sites using clean and leached funnel collectors designed to capture bulk inorganic N deposition [19]. We consider this location to represent an upper-end estimate of bulk wet inorganic N deposition to the study area as the collection site is slightly higher in elevation and receives higher orographically-driven rainfall (>9000 mm). All rain and stream water samples were field-filtered (0.7 mm glass fiber) into triple-leached polyurethane bottles and frozen until analysis [19]. Water samples were analyzed for anions via flow injection spectrophotometry (SEAL QuAATro, Merquant WI) and ion chromatography (Metrohm, Riverview FL), cations via ion chromatography and inductively coupled plasma optical emission spectrometry (Optimum 5300 DV, Perkins Elmer), 3H and 15O via laser absorption liquid-water isotope spectrometry (Model DLT-100, Los Gatos Research), and DOC and total dissolved N (TDN) via high temperature platinum combustion with a total N module (Shimadzu TOC-V) at MSU, and 15NO3 via the denitrifier method and isotope ratio mass spectrometry (UC Davis Stable Isotope Facility, Davis, CA). Deuterium-excess was calculated relative to the global meteoric water line: 3H / 1H = 8 15O / 16O Dissolved organic N (DON) was calculated as TDN – (nitrate+ammonium).

We estimated atmospheric N deposition fluxes as the geometric mean and associated quadratic error of the product of DIN (nitrate+ammonium) concentrations in rain for each event and MAP. As all of our study watersheds are ungauged, we estimated hydrologic N losses using seasonal rainfall lifts for each sampling period over 2009-2011 by applying a 50% water loss via evapotranspiration (ET, based on regional estimates for sites with similar MAP [31]) and multiplying resulting runoff by individual stream N concentrations. This approach assumes similar ET
characteristics between forest types, such as rates of canopy interception and evaporation as we currently lack data on such processes. However, we evaluated this assumption by analyzing stream samples for chemical and isotopic properties (e.g., Cl⁻ and deuterium excess) that are sensitive to watershed-level effects of dilution and evaporation. We calculated annual fluxes and associated error via quadratic error propagation using observed seasonal variation in calculated stream discharge and stream N concentrations. We estimated gas losses via denitrification using a simple steady-state isotope mass balance model [16]. We parameterized the model using observed mean bulk soil ¹⁵N, mean atmospheric deposition ¹⁵N, an isotope effect for leaching calculated by subtracting observed mean ¹⁵NO₃ in streams from mean bulk soil ¹⁵N for each forest type, and a 16‰ isotope enrichment effect for denitrification following ref 16. We estimated error in total N losses via quadratic error propagation assuming that variance in fractional gas losses varied proportionally with the observed standard deviation in ¹⁵NO₃ and combining this with the observed standard deviation in hydrologic N fluxes across all samples.

Differences in bulk soil physical, chemical and isotopic properties between forest types were tested using t-tests. Differences in levels of bioavailable soil N between forests and over time were examined using two-way ANOVA on log-transformed values followed by Tukey post hoc tests. For years in which multiple plots were sampled within watersheds (2009, 2010) initial examination (ANOVA) indicated a non-significant effect of watershed identity on soil chemistry. We therefore treat these plots as independent samples (2009, n = 6; 2010, n = 12; 2011, n = 6) in tests of forest-type effects. Stream chemical and isotopic data were analyzed using a combination of linear regression and ANOVA (all statistical tests were made using SigmaPlot). Initial analyses (ANOVA) revealed that watershed identity had no effect on intra- and inter-forest temporal and spatial patterns in stream nitrate and therefore individual watershed-years were treated as independent samples in subsequent tests.

Results and Discussion

We first examined bulk soil physical and chemical properties (bulk density, texture, pH, %C, %N or δ¹³C, and δ¹⁵N) that integrate long-term organic matter processing in the plant-soil system [32]. We found no differences in bulk density, texture, %C, %N or δ¹³C in shallow (0–15 cm) mineral soils of diverse Mora forests and only marginal (P=0.06; two-tailed t test) differences in pH (Table 1) implying similar long-term organic matter storage across these forests. We also found no differences between forests in measures of extractable soil phosphorus. Despite equivalent levels of bulk soil N, however, we found marginally (P=0.06; two-tailed t test) higher levels of δ¹⁵N in soils of diverse compared to Mora forests a pattern that implies higher N availability and microbial cycling [18] within the plant-soil N system of diverse forests.

Table 1. Physical, chemical, isotopic and biological properties of bulk mineral soils (0–15 cm).

| Soil Property | Mora | Diverse | p-value |
|---------------|------|---------|----------|
| Bulk density (g/cm³) | 1.3 (0.1) | 1.2 (0.2) | 0.32 |
| Clay (%) | 36.2 (1.8) | 31.5 (3.5) | 0.26 |
| Silt (%) | 38.2 (1.7) | 36.2 (3.1) | 0.59 |
| Sand (%) | 25.5 (2.7) | 32.3 (6.4) | 0.36 |
| pH | 3.9 (3.7–4.1) | 4.1 (3.7–4.5) | 0.06 |
| %C | 3.1 (0.5) | 2.6 (0.4) | 0.27 |
| %N | 0.3 (0.02) | 0.3 (0.03) | 0.47 |
| δ¹³C (%) | 28.9 (0.1) | 28.8 (0.3) | 0.40 |
| δ¹⁵N (%) | 4.7 (0.3) | 5.7 (0.4) | 0.06 |
| Bray’s-P (µg/g) | 2.6 (0.4) | 2.9 (0.5) | 0.52 |
| Root biomass (g/m²) | 129.4 (45.2) | 70.0 (48.8) | 0.39 |

Values are means and 1 SE (in parentheses) for watershed-level composited samples for bulk density (n=10) texture (n=5); pH (n=15); %C, %N, δ¹³C and δ¹⁵N (n=8); and Bray’s-P (n=15); and root biomass (n=5), where the sample size (n) refers to the number of plots. For pH, the mean was calculated by averaging hydrogen ion concentrations followed by log10 back transformation. The values in parentheses are quadratic ranges. P-values are the results of two-sided t-tests. doi:10.1371/journal.pone.0070491.t001

Whitney Rank Sum Test). These differences in the organic layer would also predict low nitrate accumulation in deeper soils. Accordingly, we found that mineral soils of diverse forests contained much higher nitrate levels, independent of season, that were characteristic of N-rich ecosystems [18], while nitrate in Mora forests was consistently low (Fig. 1B, 1E). Ammonium was always the dominant form of mineral N in Mora soils, a pattern common to N-limited temperate forests [33]. In contrast, high nitrate: ammonium ratios generally occur under conditions of high N availability where nitrate can accumulate [34]. During the wet season, ammonium was the dominant form of available N in all soils and resulted in low nitrate: ammonium ratios, particularly in organic horizons (Fig. 1C). Across all dry season samples, however, nitrate: ammonium ratios were >10–100 times higher in diverse compared to Mora forests (Fig. 1F). Concentrations of inorganic N in Mora forests remained unchanged across wet and dry seasons over a three year period. In contrast, over the same seasonal sequence diverse forests showed a significant decrease in ammonium, a six-fold increase in nitrate, and a fifty-fold increase in nitrate: ammonium ratios.

This difference could not be explained by differences in soil moisture (Fig. S1) suggesting that low bioavailable N in Mora forests does not result from differential availability of water between forest types but, rather, differential sensitivity of the N cycle to rainfall. Low and unvarying levels of bioavailable N indicate remarkable stability of the internal Mora N cycle despite large seasonal variation in rainfall. Large seasonal variation in diverse forests on the other hand is consistent with an internal N cycle sensitive to moisture and high microbial nitrate production [35] leading to high mineral N losses from the plant-soil ecosystem [18].

We tested this prediction by analyzing dissolved N in small watershed streams. Across all forests and sampling periods, our analysis revealed ~2-fold lower concentrations of dissolved inorganic N (DIN) in streams draining Mora forests than in diverse forests. Nitrate was significantly lower (P<0.001, two-way ANOVA) in Mora forests compared to its diverse neighbors.

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Nitrogen Cycling in Monodominant Tropical Forest
while concentrations of ammonium were low across all forests (average ± SEM, 5 ± 0.4 μg N/L, n = 78) and did not differ between forest types. Stream nitrate concentrations in our diverse forests are similar to those reported for other N-rich tropical forests while those in Mora forests are at the low end of observations for forests of similar climate worldwide [15,17].

Several physical factors external to the plant-soil system could give rise to lower nitrate concentrations in streams draining Mora forests. For example, the propensity for N losses could change systematically with elevation or slope. Diverse forests occupy a broader topographic range than Mora, but we observed constant between-forest differences and within-forest variance in nitrate

Figure 1. Soil N availability in monodominant (Mora) and diverse rain forests in Trinidad. (A–C) Box plots (median, 5th and 95th percentiles) of ammonium, nitrate, and nitrate: ammonium ratios in surface organic soils near focal trees collected during the wet season (2012). Levels of ammonium (P = 0.007, two-tailed t test) and nitrate (P = 0.073, two-tailed t test) were lower in soils of Mora forests (white symbols) than in diverse forests (grey symbols) but nitrate: ammonium ratios did not differ (P = 0.85, two-tailed t test). (D, E) Box plots (median, 5th and 95th percentiles) showing levels of ammonium and nitrate in mineral (0–15 cm) soils during dry (2009 and 2010) and wet (2012) seasons. Soils were not measured for inorganic N in 2011. Across years ammonium varied significantly (P = 0.017 for year effect) but did not differ consistently between forest types (P = 0.26 for type effect) but the effect of year depended on forest type (P < 0.001 for interaction term, two-way ANOVA). In contrast, nitrate was consistently lower in Mora than diverse forests across years and did not change through time whereas nitrate in diverse forests showed increases during dry seasons (P < 0.001 for type effect, P = 0.029 for year effect, and P = 0.009 for interaction term, two-way ANOVA). This resulted in low and stable nitrate: ammonium ratios in Mora compared to diverse forests (P < 0.001 for all effects, two-way ANOVA).

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Figure 2. Spatial and temporal distribution of nitrate and d-excess in small watershed streams. (A) Across all samples stream nitrate did not change as a function of watershed elevation (median of ridge top and watershed outlet elevations) within Mora (white symbols, n = 38) or diverse (grey symbols, n = 40) watersheds or across all watersheds combined (P > 0.2 for all comparisons, linear regression and 95% CI). (B) Mean (± SEM) stream nitrate concentrations over the 2009–2011 period in Mora (n = 5, 13, 6, and 14) and diverse (n = 12, 9, 6, and 12) watersheds. Nitrate in stream waters was consistently higher and more variable in diverse than Mora forests across years (P < 0.001 for type effect, P < 0.001 for year effect, and P = 0.028 for interaction term, two-way ANOVA). Across the drought to post-drought sequence nitrate in Mora forest remained unchanged (P > 0.57) while nitrate in diverse forests increased significantly (P < 0.01). (C) Across all samples d-excess did not change as a function of elevation within Mora (white symbols, n = 32) or diverse (grey symbols, n = 30) watersheds or across all watersheds combined (P > 0.16 for all comparisons, linear regression and 95% CI). (D) Mean (± SEM) d-excess in stream waters varied significantly and synchronously across time (P < 0.001 for year effect) but did not differ between forest types across or within years (P = 0.104 for type effect, P = 0.933 for interaction term, two-way ANOVA).

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Nitrogen Cycling in Monodominant Tropical Forest
across all elevations (Fig. 2A). We also considered whether differences in watershed hydrologic fluxes could explain the nitrate pattern. Diluted nitrate concentrations could result from higher runoff or lower rates of evaporation in Mora forests. In the former case we would predict proportional dilution of more biologically conservative solutes such as Cl\(^-\), SO\(_4^{2-}\), or base cations. However, we found no differences in concentrations of cations or SO\(_4^{2-}\) between forest types, suggesting similar runoff characteristics among watersheds, and only marginally higher levels of Cl\(^-\) in Mora forests (Fig. 82), a pattern opposite to that predicted if dilution explained the nitrate pattern. Similarly, we evaluated whether evaporation differences among watersheds might control nitrate concentrations by examining distributions of stable water isotopes in stream water. Across all samples, deuterium excess, which traces local water sources and evaporation [36], did not change with elevation nor differ between forest types over time (Fig. 2C, 2D). We conclude that differences in topography, runoff, and evaporation are unlikely to explain the inter-forest differences in nitrate losses observed in these catchments.

We next examined the sensitivity of stream nitrate to inter-annual variation in rainfall. Figure 2B illustrates that nitrate remained lower in Mora than diverse forests prior to, over the course of, and following a severe drought (Dec. 2009–Jun. 2010). Over the drought-to-wet season sequence, stream nitrate in diverse forests showed a dramatic and sustained two-fold increase, while nitrate in Mora forests remained unchanged. Large increases in nitrate concentration with increasing discharge are typically observed in ecosystems with high internal nitrate production and nitrate accumulation in soils [18]. Consistently higher nitrate in soils of diverse than Mora forests corroborates that this difference is an internally generated and temporally stable feature of the N cycle of these ecosystems, at least over the period of our study.

Using N concentrations from all streams and basin-scale water fluxes we estimate that exports of DIN are \(~2.6\) kg N ha\(^{-1}\) yr\(^{-1}\) (2.2–3.1 kg N ha\(^{-1}\) yr\(^{-1}\), quadratic range) from diverse forests and \(~1.1\) kg N ha\(^{-1}\) yr\(^{-1}\) (1.0–1.3 kg N ha\(^{-1}\) yr\(^{-1}\)) from Mora forests. To sustain such differences in hydrologic N losses from these old-growth forests requires differences in other pathways of N loss or differences in external inputs of N [14]. As these forests are all old-growth and have acid soils, N losses from fire or volatilization are unlikely vectors. Instead, we focus on losses via dissolved organic N (DON) leaching and gaseous N via denitrification since the former has been shown to sustain N limitation in old forests [14] and the latter can be significant in N-rich tropical forests [16].

Across all our study watersheds we found low levels of stream dissolved organic carbon (DOC, 0.1–1.1 mg/L) and DON but that levels of DON were on average higher \((P<0.01;\) two-tailed t test) in streams draining Mora (Fig. 3A). However, this difference could not balance the difference in dissolved N losses between forests (accounting for \(<5–20%\) of total dissolved N losses). Alternatively, elevated gaseous losses via denitrification could explain the observed imbalance. We examined this possibility by first analyzing the natural abundance \(\delta^{15}N\) distributions of stream water nitrate. We observed 1) \(\delta^{15}NO_3\) was significantly higher in diverse forest streams than Mora streams (Fig. 3B), and 2) \(\delta^{15}NO_3\) became increasingly enriched with increasing nitrate concentrations (Fig. 3C), a pattern expected from microbial denitrification [18] and consistent with enriched soil \(^{15}N\) in diverse forests soils. Combined, these results suggest an ecosystem-level signature of denitrification opposite that expected if denitrification accounted for the observed N imbalance, supporting our inference that total mineral N losses are greater from diverse forests compared to Mora forests.

We next examined the contribution of gaseous losses to total N losses from these forests and how these compare to abiotic N inputs via atmospheric deposition. We achieved this in two ways: 1) direct measures of DIN and \(\delta^{15}NO_3\) in rainwater; and 2) N isotope mass balance. We combined DIN concentrations from 22 rain events sampled over four years with mean annual precipitation to estimate bulk inorganic N deposition inputs (2.7 kg N ha\(^{-1}\) yr\(^{-1}\); quadratic range: 1.3–3.6 kg N ha\(^{-1}\) yr\(^{-1}\)), consistent with previous field [18] and model [37] estimates of reactive N deposition in the region. We estimated N gas losses using an isotope mass balance model [16] parameterized with direct measures of \(\delta^{15}N\) in bulk soils (Table 1), \(\delta^{15}NO_3\) in stream water (Fig. 3B and 3C) and precipitation (average \(\pm\) SEM, 1.4\(\pm\)0.7\%o, \(a=1\)), and a mean fractionation factor for soils from the literature [16]. This approach yielded denitrification rates equal to 49% and 44% of total N losses from diverse and Mora forests, respectively. Combining gaseous losses with hydrologic N fluxes, we estimate that total inorganic N losses are \(~5.1\) kg N ha\(^{-1}\) yr\(^{-1}\) (quadratic range: 4.3–5.9 kg N ha\(^{-1}\) yr\(^{-1}\)) from diverse forests and \(~2.0\) kg N ha\(^{-1}\) yr\(^{-1}\) (quadratic range: 1.7–2.3 kg N ha\(^{-1}\) yr\(^{-1}\)) from Mora forests, resulting in inorganic N loss: input ratios \(>1\) in diverse forests and \(<1\) in Mora forests. Importantly, we find that leaching losses approximately balance inputs in diverse but not Mora forests even if we exclude gaseous losses entirely. We conclude that bioavailable N losses likely exceed abiotic inputs in diverse forests but are unlikely to in Mora forests (Table 2). This implies a fundamental difference in the N cycle of these forests, as the former pattern suggests the lack of N limitation, while the latter suggests its possibility [14].

For these differences in N losses to be sustained, as our evidence suggests, requires that external inputs of N to diverse forests exceed those to Mora forests. That is, assuming that soil N is near steady state our finding of no differences in bulk soil %N (Table 1) but large and sustained differences in N losses between forests means that N inputs likely differ. This could result from higher rates of biological N\(_2\) fixation associated with diverse forests [15]. In addition to widespread asymbiotic N\(_2\) fixation in tropical forests in general [38], our study area harbors leguminous tree species (e.g., *Pentaclethra macroloba*, *Inga sp.*) known to fix large amounts of N [15,39], but which are less abundant in Mora forest [5,25]. Given its extreme canopy, understory and litter-layer dominance, it is possible that Mora acts to exclude these potential N fixers despite lower soil N availability. In contrast to symbiotic N\(_2\) fixation, however, the C-rich and low inorganic N root-litter layer of Mora forests would seem to favor asymbiotic N\(_2\) fixers [15] but we currently lack data on this process. At the same time, if this external input is significant in Mora forests, our results suggest that biological cycling of this new N must be extremely efficient and/or or that total N pools are still growing or that higher DON losses balance this input given that mineral N pools and losses are small. Lower N supply should in turn favor plant traits which tightly conserve N in the plant-soil system.

*Mora* has low foliar and litter N concentrations and low litterfall N fluxes [25] relative to other tropical trees globally [40,41] and shows significant foliar resorption of N [25]. Slow decomposition rates cause *Mora* litter to accumulate and it has been speculated that it is from this soil layer that surface roots of *Mora* procure their nutrient requirements [25]. Nitrate levels in *Mora* soils are exceptionally low and similar to those found in N-limited temperate forests [33]. Low soil nitrate has also been found in the monodominant *Gilbertiodendron dewevrei* (Canalipseudinae) forests of Africa [9]. Such low nitrate relative to other tropical forests could result from preferential plant drawdown of ammonium (by reducing the ammonium pool available to microbial nitrifiers) or
nitrate [42] and/or a variety of microbial processes that act to lower nitrate production [43–45]. These mechanisms are consistent with the theoretical prediction that relatively high plant preference for ammonium and ability to inhibit nitrification can allow for invasion of and resistance to invasion by, plants without these traits [42]. These mechanisms could be examined via a combination of reciprocal litter and seedling transplant experiments and N fertilization and 15N-tracer additions. Irrespective of the plant-microbial mechanism/s underlying low soil nitrate, however, we show that low soil nitrate associated with monodominance has ecosystem-level consequences for patterns of N loss. This results from the fact that nitrate is both highly mobile in soils and is also the substrate for microbial denitrification. Thus lower nitrate in soils should translate directly to smaller hydrologic and gaseous N losses at the ecosystem level [15,18], which is what we observed.

Though it is reasonable that known functional traits of Mora (canopy dominance, shade tolerance, deep litter) would provide a competitive advantage in some environments, it remains unknown how Mora comes to dominate otherwise diverse landscapes and the degree to which this depends on initial conditions, including characteristics of the physical environment, and disturbance and invasion dynamics. Here, we find that Mora dominance late in forest succession is associated with strong reductions in levels of bioavailable N in the plant-soil system and losses at the ecosystem level. This differentiates Mora from other trait-driven N cycles in which tree species increase ecosystem N supply through symbiotic N2 fixation following disturbance or invasion [21,22]. Curiously, the majority of monodominant taxa worldwide are Caesalpinioideae legumes [10], a group in which symbiotic N2-fixation is rare [46]. In many diverse tropical forests, N2 fixation by legumes may drive post-disturbance N accumulation [34] but this input slows via negative plant-soil feedback as available N pools increase [15,39]. Positive or self-reinforcing feedback, on the other hand, tends to destabilize plant-nutrient systems [47]. For the Mora system, if positive N feedback is operating, progressive draw-down of available N could ultimately exacerbate N limitation and could both drive selection toward increasing N use efficiency but also increase system sensitivity to perturbations that alter the competitive environment. We suggest that forest monodominance in the tropics provides a promising model system for exploring the organization of the tropical N cycle in particular and higher-order consequences of species functional traits in general.

### Supporting Information

**Figure S1** Concentrations of soil ammonium and nitrate as a function of gravimetric soil moisture. A, B) Concentrations of ammonium and nitrate in organic horizons of forest soils during the wet season (2012). C, D) Concentrations of ammonium and nitrate in mineral (0–15 cm) soils during dry (2010; triangles) and wet (2012; circles) seasons. Variation in soil moisture did not explain any significant (P>0.20) within- or between-forest variation in soil N. (TIF)

**Figure S2** Concentrations of anions and cations in small watershed streams of diverse and Mora forests of Trinidad. (A) Box plots (median, 5th and 95th percentiles and outliers) of stream chloride in diverse (grey symbols n=18) and Mora (white symbols, n=15) forests. Stream chloride decreased strongly and synchronously between dry (2010) and wet (2011) years (P<0.001 for year effect) and differed between forest types (P=0.013 for type effect) with no significant year-type interaction (P=0.139 for interaction term, df=26, two-way ANOVA). Slightly higher chloride in Mora forest is opposite to that expected if dilution explained lower stream nitrate, is not consistent with other ions and water isotopes and could be explained by slightly closer proximity to the ocean. (B) Box plots (median, 5th and 95th percentiles) of stream sulphate in diverse (n=18) and Mora (n=15) forests. Sulphate also decreased significantly and synchronously from dry to wet seasons (P<0.001 for year effect) but did not differ between forest types across or within years (P=0.096 for type
effect, $P = 0.611$ for interaction term, df = 26, two-way ANOVA). (C) Box plots (median, 5th and 95th percentiles) of the sum of stream cations ($K^+$, $Na^+$, $Ca^{2+}$, $Mg^{2+}$). Cations showed significant and synchronous increases from 2009 to 2010 ($P = 0.001$ for year effect) and no differences between forest types across or within years ($P = 0.227$ for type effect, $P = 0.116$ for interaction term, df = 29, two-way ANOVA). (THF)

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Author Contributions

Conceived and designed the experiments: ENJB SAT. Performed the experiments: ENJB SAT. Analyzed the data: ENJB. Wrote the paper: ENJB SAT.

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