Ammonium as a Driving Force of Plant Diversity and Ecosystem Functioning: Observations Based on 5 Years’ Manipulation of N Dose and Form in a Mediterranean Ecosystem

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Abstract

Enhanced nitrogen (N) availability is one of the main drivers of biodiversity loss and degradation of ecosystem functions. However, in very nutrient-poor ecosystems, enhanced N input can, in the short-term, promote diversity. Mediterranean Basin ecosystems are nutrient-limited biodiversity hotspots, but no information is available on their medium- or long-term responses to enhanced N input. Since 2007, we have been manipulating the form and dose of available N in a Mediterranean Basin maquis in south-western Europe that has low ambient N deposition (<4 kg N ha⁻¹ yr⁻¹) and low soil N content (0.1%). N availability was modified by the addition of 40 kg N ha⁻¹ yr⁻¹ as a 1:1 NH₄Cl to (NH₄)₂SO₄ mixture, and 40 and 80 kg N ha⁻¹ yr⁻¹ as NH₄NO₃. Over the following 5 years, the impacts on plant composition and diversity (richness and evenness) and some ecosystem characteristics (soil extractable N and organic matter, aboveground biomass and % of bare soil) were assessed. Plant species richness increased with enhanced N input and was more related to ammonium than to nitrate. Exposure to 40 kg NH₄⁺-N ha⁻¹ yr⁻¹ (alone and with nitrate) enhanced plant richness, but did not increase aboveground biomass; soil extractable N even increased under 80 kg NH₄NO₃-N ha⁻¹ yr⁻¹ and the % of bare soil increased under 40 kg NH₄⁺-N ha⁻¹ yr⁻¹. The treatment containing less ammonium, 40 kg NH₄NO₃-N ha⁻¹ yr⁻¹, did not enhance plant diversity but promoted aboveground biomass and reduced the % of bare soil. Data suggest that enhanced NH₄⁺ availability affects the structure of the maquis, which may promote soil erosion and N leakage, whereas enhanced NO₃⁻ availability leads to biomass accumulation which may increase the fire risk. These observations are relevant for land use management in biodiverse and fragmented ecosystems such as the maquis, especially in conservation areas.

Introduction

High biodiversity can stabilize ecosystems through functional complementarities, which can buffer the impacts of environmental change [1,2]. Biodiversity and ecosystem functions are influenced by several drivers (e.g. land use change, increased nutrient availability), but ecosystem responses (and mechanisms) to those drivers remain unclear [3], especially responses of ecosystems under persistent anthropogenic influence such as enhanced nitrogen (N) deposition.

Enhanced N availability has been acknowledged as a global and increasing threat to biodiversity [4–7] and ecosystem function [8,9]. However, most of our knowledge of the impacts of increased N availability on ecosystems comes from northern Europe and America [6]. Mediterranean-type ecosystems appear on the ‘neglected ecosystems list’ [6,10] despite being a global conservation priority [11,12], rivalling tropical rainforests [12,13]. In fact, not much is known about the impacts of increased N availability on Mediterranean-type ecosystems, other than those found in California [10,14]. Ecosystems in the Mediterranean Basin that have experienced intensive human development and impact for millennia [15], and where N deposition is expected to increase threefold by 2050 [5,16] are particularly deserving of study.

The most distinctive features of Mediterranean-type ecosystems likely to influence responses to increased N availability are: climate (highly seasonal, with warm dry summers that contrast with cool rainy winters); soils (low nutrient levels and organic matter and high contents of bases such as carbonates); dominance of dry N deposition; asynchrony between N availability and biological activity [10]; and spatial and temporal heterogeneity [17]. On the basis of these distinguishing features, it seems likely that Mediterranean-type ecosystems could respond rather differently from north temperate ones. Even within Mediterranean-type
ecosystems, differences in soil fertility [13] and phosphorus (P) availability in particular can undermine extrapolations from Californian ecosystems to those in the Mediterranean Basin [10]. For these reasons, in 2007 an N-manipulation (dose and form) field experiment was established in a severely nutrient-limited Mediterranean Basin maquis. Contrasting with most studies (north temperate systems [6-9] and Mediterranean Basin [18,19]), within one year plant richness increased and changes in plant community composition were observed [20]. Recently, Forest et al. [3] provided data supporting the hypothesis that long-term impacts of N enrichment on ecosystem functioning depend strongly on biodiversity changes, especially non-random changes in species composition. As a result, our observation raises key questions:

(i) Is this N-driven increase in plant richness transient? This is especially relevant since N deposition in most European ecosystems has already reached a threshold, beyond which diversity has declined [21]. Accordingly, given our N doses, we expected that after 5 years, plant diversity would have stabilized or be in decline.

(ii) Are there consistent plant responders to increased N availability in Mediterranean maquis? We hypothesized that the changes in plant cover and species composition along the ecological succession [22] may confound the identification of consistent plant responders to enhanced N availability.

(iii) Are the N-driven changes in diversity linked with plant productivity? Experiments that directly manipulate species diversity often report a positive impact on productivity, whereas observations of natural communities reveal more complex relationships [2], and nutrient additions generally increase productivity but decrease diversity [23]. In our case, because we expected that the N-benefited species would be small, short-lived plants [24,25] that contribute little to ecosystem functioning, even if plant richness continued to increase, we did not expect an increase in productivity.

Inherent to all these questions is the need to understand whether the form of N matters more than the dose. This is especially relevant since the co-existing plant species occupy distinct N niches, with ammonium (NH$_4^+$) availability as a determinant [17,26], so that the plant’s response to enhanced N availability may vary according to the N form irrespective of the dose.

We report 5 years of results from an N-manipulation field experiment, focusing on N-driven changes among the plant community and soil characteristics in a Mediterranean Basin maquis. Specifically, we focused on the following ecosystem features: vascular plant community (composition, richness and evenness), aboveground biomass (standing biomass and litter production), soil extractable N (NH$_4^+$ and nitrate – NO$_3^-$) and organic matter, and percentage of bare soil.

**Materials and Methods**

**Study site**

We are grateful to Arrábida Natural Park for making the experimental site available and allowing the N manipulation experiment to which this paper refers. The study site (38°29'N - 9°11'W) is located in Serra da Arrábida in the Arrábida Natural Park, Portugal (a Natura 2000 site - PTCON0010 Arrábida/ Espichel), which is within the sub-humid thermomediterranean bioclimatic domain [27]. According to records (1971–2000 - Instituto Nacional de Meteorologia e Geofisica), mean annual precipitation is 730 mm; mean maximum temperature, 27.8°C (August); and mean minimum temperature, 8.1°C (January). Over the experimental period (2007–2011), mean annual precipitation was 870 mm (±231 mm, SE); mean maximum temperature, 21.6°C (±0.7°C, SE); and mean minimum temperature, 12.9°C (±0.5°C, SE). Total monthly precipitation and mean monthly temperatures over the experimental period are shown in Figure 1.

The site is located on a southeast-facing slope (5%) at 130 m altitude, which is protected from public access and has not been managed in recent decades. The soils of Serra da Arrábida are classified, according to the FAO system, as calcic rhodo-chromic luvisols and calcareous cambisols [28]. The soil is skeletal (topsoil layer of approximately 15 cm) with a bulk density of 1.3 g cm$^{-3}$. Silt predominates (50%), while sand and clay contents are 32% and 18% (silt-sand-loam texture).

Mediterranean maquis vegetation dominates the site, comprising closed vegetation: mainly shrubs with few annuals and some geophytes and normally with some trees, some of which may be in shrub form (Eunis class F 5.2 – Mediterranean maquis). The standing community developed after a fire event in summer 2003, four years before the first N addition of this experiment. The dominant plant species was a Gistaceae, Cistus ladanifer L. [20,29]. Other abundant plant species were Erica scoparia L. (Ericaceae), Calluna vulgaris (L.) Hull (Ericaceae), Genista truncatissimus Bros. (Fabaceae), Ulex densiflorus Welw. ex Webb (Fabaceae), Dictamnus albus L. (Asteraceae) and Myrtus communis L. (Myrtaceae). Herbaceous species, of which many were annual plants, comprised ≈10% of the total plant cover [20].

**Experimental design and fertilization schedule**

During the experimental period (2007–2011), estimated background N deposition ranged between 2.9 kg ha$^{-1}$ yr$^{-1}$ (1.8 kg NO$_3^-$+1.1 kg NH$_4^+$) in 2008 and 3.8 kg ha$^{-1}$ yr$^{-1}$ (2.2 kg NO$_3^-$+1.6 kg NH$_4^+$) in 2010. These values were estimated based on the model used by the European Monitoring and Evaluation Programme (grid location: x = 54 and y = 4 - http://www.epem.int/mscw/index_mscw.html). The chosen N doses were high enough to simulate ‘worst case’ scenarios of N enrichment in this type of habitat, but lower than the N deposition reported for highly N polluted areas in Mediterranean-type ecosystems [30,31]. The N forms mimicked the most likely N pollution scenarios within the Mediterranean Basin (e.g. agricultural sources alone or combined with urban/industrial sources). Control plots received no added N, while there were three N treatments: 40A received 40 kg NH$_4^+$-N ha$^{-1}$ yr$^{-1}$ as a 1:1 mixture of NH$_4$Cl and (NH$_4$)$_2$SO$_4$; 40AN received 20 kg NH$_4^+$-N ha$^{-1}$ yr$^{-1}$ and 20 kg NO$_3^-$-N ha$^{-1}$ yr$^{-1}$ as NH$_4$NO$_3$; and 80AN received 40 kg NH$_4^+$-N ha$^{-1}$ yr$^{-1}$ and 40 kg NO$_3^-$-N ha$^{-1}$ yr$^{-1}$ as NH$_4$NO$_3$. Thus the 40A and 40AN treatments provided the same N dose, while 40A and 80AN provided the same NH$_4^+$ dose. To prevent N ‘contamination’ through runoff from the N-plots, the experimental plots were randomly distributed in three rows across the slope, with the controls being located in the top row.

Beginning in January 2007, the dry N salts were homogenously added, by hand, sprinkled over the soil surface, in three equal applications over the year: mid-autumn/winter, spring and summer (Fig. 1). Each treatment was replicated three times (3 plots of 400 m$^2$ each). To restrict boundary effects and dilution processes, all measurements, analyses and sample collection were performed within the central 100 m$^2$ square.

**Plant diversity and % of bare soil**

The composition of the vascular plant community was assessed in June 2007, May 2008 [20] and June 2011 within one 5 x 5 m
square per experimental plot (within the internal 100 m²). Percentages of species cover calculated from the total projected crown area and of bare soil (as a measure of erosion potential) were recorded. Plant species observed in the three community assessments are grouped by life form in Table S1 using the Flora Digital de Portugal database (http://jb.utad.pt/pesquisa). This classification was considered more informative than others (e.g. by forb, grass, shrub and tree) because it provides some clues to plant size and life cycle. From the vascular plant community assessments in 2007 and 2011, it was possible to calculate plant richness and evenness for the first and fifth springs of the experiment.

Soil extractable N and organic matter

Soil inorganic N pools were measured to estimate soil N retention (the lower the soil inorganic N pools, the higher the N retention and vice versa), and organic matter was measured as a proxy for belowground C sequestration. Soil was sampled from the four corners and the centre of the central 100 m² square of each plot. Soil samples (2 cm diameter and 15 cm depth) were removed, sieved (2 mm) and stored at 4°C until analysis. Sampling took place in May 2007 (first spring) and 2011 (fifth spring of the N additions). Individual soil samples (five per plot) were analysed for nitrate (NO₃⁻-N [33]), ammonium (NH₄⁺-N [34]), pH (H₂O extract) and organic matter [20]. Soil inorganic N (Nₐ) was the sum of the water-extracted NH₄⁺ and NO₃⁻. NO₃⁻, NH₄⁺ and Nₐ were expressed as μg N per gram of dry soil. Bulk soil samples (equal mixtures of the five soil samples from each experimental plot) were dried (at 60°C until constant mass), ground (MM 2000) and used for quantifying total soil N and C, and hence the C/N ratio, by dry combustion using an elemental analyser (EuroVector, Italy).

Derived variables and statistics

The cumulative N, NH₄⁺ and NO₃⁻ load (estimated N deposition +N additions) at each plant assessment were calculated using the above-mentioned EMEP estimates and assuming that N, NH₄, and NO₃ were deposited homogeneously over the year (Fig. 3).

Summary statistics of soil properties, plant species and community responses (richness, evenness, cover, biomass and litterfall) of the various N additions were compared. Two-way ANOVA was applied to determine if there were significant interactions between time and treatment for soil and plant variables. Differences per treatment in biomass and litterfall were analysed by a one-way ANOVA. Both types of ANOVA were followed by a Bonferroni test (p<0.05 or p<0.1), or by a Games-Howell test whenever homogeneity of variances was not confirmed by the Levene’s test. Between treatment differences for change in plant cover were analysed by a one-way ANOVA (followed by a Bonferroni test p<0.1) and a Kruskal-Wallis test (p<0.01) for normal and non-normal samples respectively (Table S1). Linear correlations between plant richness and cumulative N, NH₄⁺ and NO₃⁻ were also studied (Pearson’s correlations). Correlation between plant richness and cumulative NH₄⁺ and that between plant richness and cumulative NO₃⁻ were compared using the
Between plant richness and cumulative NO$_3^-$ relation to the control) in treatments receiving 40 kg NH$_4^+$. During the same period, plant evenness decreased with time, decreasing least in the treatments (two-way ANOVA; Different letters refer to statistically significant differences between treatments (a) and evenness (b). Community assessments were performed in the treatments (Control, 40A, 40AN and 80AN), in terms of plant richness (Fig. 2). Richness and composition changed the most (p=0.05) in relation to the control) in treatments receiving 40 kg NH$_4^+$.-N ha$^{-1}$ yr$^{-1}$ (40A and 80AN – Fig. 2 and Table S1). In contrast, between 2007 and 2011, plant species richness (Fig. 2) did not change in the control or 40AN plots. Linear, positive and significant correlations were found between plant richness and cumulative N, NH$_4^+$ and NO$_3^-$ (Fig. 3). However, the correlation between plant richness and cumulative NH$_4^+$ addition was higher than that between plant richness and cumulative NO$_3^-$.

Impacts of the N treatments on plant diversity
As usual in the Mediterranean region, the annual rainfall was highly variable (ranging from 513 mm in 2007 and 1541 mm in 2010), with rainfall being greatest during the last two rainy seasons of the experiment (2009/2010 and 2010/2011) (Fig. 1). Initially (spring 2007), richness and evenness were similar in all treatments (Fig. 2). Between treatment differences were, however, evident after one year [20], and remained throughout (Table S1 and Fig. 2). Richness and composition changed the most (p=0.05 in relation to the control) in treatments receiving 40 kg NH$_4^+$.N ha$^{-1}$ yr$^{-1}$ (40A and 80AN – Fig. 2 and Table S1). In contrast, between 2007 and 2011, plant species richness (Fig. 2-a) did not change in the control or 40AN plots. Linear, positive and significant correlations were found between plant richness and cumulative N, NH$_4^+$ and NO$_3^-$ (Fig. 3). However, the correlation between plant richness and cumulative NH$_4^+$ addition was higher than that between plant richness and cumulative NO$_3^-$.

Impacts of N on ecosystem processes
N additions did not affect total N or C concentrations after one and five years (Tables 2 and S3). Soils contained very little N (0.1%) or C (~2%), resulting in a C/N ratio <20. N treatments did increase soil available N and organic matter. In relation to the N dose, control plots contained significantly lower concentrations than the 80AN treatment plots, and plots receiving 40 kg N ha$^{-1}$ yr$^{-1}$ (40A and 40AN) showed intermediate levels. Soil NO$_3^-$ and soil inorganic N (these two variables were highly correlated, r=0.98, p<0.001) increased over time. The soil also acidified over the course of the experiment, independent of treatment (Tables 2 and S3).

The addition of 40AN increased aboveground standing biomass over the control (Fig. 4-a). When estimated leaf litter (Fig. 4-b) produced between 2007 and 2011 is taken into account, aboveground biomass produced by the 40AN plants also exceeded controls (Fig. 4-c). Initially control plots had most bare ground (Fig. 5), but by the fifth spring of the experiment, differences between treatments were visible: addition of 40AN reduced the area of bare ground whereas 40A increased it.

Discussion
N-driven enhanced plant richness: is the impact transient?
To our knowledge, this is the first integrated study of the impacts of different N doses and forms on a Mediterranean Basin.
ecosystem. In this post-fire successional ecosystem, addition of N continued to promote vascular plant richness, even after 5 years. Assuming that, in this ecosystem, N availability is a severe environmental limitation, the N-driven increase in richness may be explained by the revised Grime’s humped-back model [37,38]. The richness increment would reflect the initial alleviation of the stress condition (N limitation), allowing the coexistence of the characteristic site species [35,39] with the incoming exploitative ones [40,41]. The sustained increase in plant richness during the 5 years of N additions may be due to four non-exclusive causes:

1. The difference in life span between the N-benefited (with short life cycles, including annuals) and the N-affected species (perennials), which would allow the detection of the incomers but only drastic impacts on the losers;

Figure 3. Relation between cumulative N and plant richness. Pearson’s correlations between plant richness (number of vascular plant species per 25 m²) and cumulative N (a), NH₄⁺ (b) and NO₃⁻ (c) over the experiment, accounting for estimated background deposition (EMEP - see material and methods). Correlation between plant richness and cumulative NH₄⁺ differed significantly from that between plant richness and cumulative NO₃⁻ (Steiger’s Z test \(p<0.05\)). Symbols represent the mean \(n=3\) experimental plots per treatment and per year) ±SE, but correlations were based on the individual values \(n=36\).

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The observed reduction in *C. ladanifer*’s cover [29] in this experiment probably resulted from its sensitivity to increased NH$_4^+$ availability [17,26], even in the presence of high NO$_3^-$ availability [26]. The reduction of its canopy will have greatly increased light availability near the soil surface, allowing ‘exploitative’ species to establish themselves, increasing plant richness. In the presence of NO$_3^-$ but not too much NH$_4^+$ (40AN treatment), the overshadowing by *C. ladanifer* is likely to have excluded these small exploitative species. Keeping in mind that the range in NO$_3^-$ in our experiment is smaller than that of NH$_4^+$, the sensitivity of *C. ladanifer* to increased NH$_4^+$ availability would explain why plant richness was more related to the cumulative ammonium inputs than to the cumulative nitrate inputs. This NH$_4^+$-driven increase in plant richness is unlikely to be observed when ecosystems are, or become, dominated by late successional species as these are more NH$_4^+$ tolerant [17,26]. However, it is possible that climbing plants already present in the community (*e.g.* *Lonicera implexa*, *Rubia peregrina*, *Rubus ulmifolius*, etc.) may become dominant in later stages of succession, as in tropical ecosystems [6]. In conclusion, even though there are no data on the impacts of long-term enhanced N availability on Mediterranean Basin ecosystems, we consider that the N-driven increase in plant richness is unlikely to be explained by increased plant richness as it did not increase in the control or 40AN plots;

A large part of the added N is being lost from the ecosystem. This is an obvious hypothesis but one for which there is no supporting data (*i.e.* runoff, nitrate leaching, ammonia volatilization and denitrification were not measured). Although NO$_3^-$ leaching is considered to be an early indicator of N ‘saturation’ [8] in Mediterranean ecosystems, it has only been measured in Californian chaparral [43] and even then the N loss was less than expected due to immobilisation by the rapidly growing vegetation [44]. N accumulation in vegetation that system was estimated to be 73 kg N ha$^{-1}$ yr$^{-1}$ [45], which is similar to our high N dose (80 kg N ha$^{-1}$ yr$^{-1}$). To quantify how much N the system could potentially ‘process’ before entering the descending phase of the humped model, longer-term studies are necessary.

**Table 1.** Plant species potentially indicative of the N dose and form.

<table>
<thead>
<tr>
<th>N dose</th>
<th>Benefitted</th>
<th>Affected</th>
</tr>
</thead>
<tbody>
<tr>
<td>40 kg N ha$^{-1}$ yr$^{-1}$</td>
<td>- Carina corymbosa</td>
<td>- Gladiolus illyricus ssp reuteri</td>
</tr>
<tr>
<td>80 kg N ha$^{-1}$ yr$^{-1}$</td>
<td>- Salvia scclareoides</td>
<td>- Cistus ladanifer</td>
</tr>
<tr>
<td>Affected</td>
<td>- Asphodelus ramosus</td>
<td>- Blackstonia perforata</td>
</tr>
<tr>
<td></td>
<td>- Dactylis glomerata</td>
<td>- Dactylis glomerata</td>
</tr>
</tbody>
</table>

Plant species that responded consistently (over 5 years of N addition treatments) to the N dose and/or form (Tables S1 and S2).

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**Table 2.** Impact of the N treatments on soil properties.

<table>
<thead>
<tr>
<th>Soil properties</th>
<th>Control</th>
<th>40A</th>
<th>40AN</th>
<th>80AN</th>
</tr>
</thead>
<tbody>
<tr>
<td>N (%) 2007</td>
<td>0.1±0.0</td>
<td>0.1±0.0</td>
<td>0.1±0.0</td>
<td>0.1±0.0</td>
</tr>
<tr>
<td>2011</td>
<td>0.1±0.0</td>
<td>0.1±0.0</td>
<td>0.1±0.0</td>
<td>0.1±0.0</td>
</tr>
<tr>
<td>C (%) 2007</td>
<td>1.6±0.2</td>
<td>1.6±0.3</td>
<td>1.8±0.1</td>
<td>2.0±0.6</td>
</tr>
<tr>
<td>2011</td>
<td>1.8±0.1</td>
<td>1.9±0.2</td>
<td>2.1±0.1</td>
<td>2.2±0.3</td>
</tr>
<tr>
<td>C/N ratio 2007</td>
<td>18.5±0.8</td>
<td>16.6±0.1</td>
<td>17.4±1.9</td>
<td>16.6±1.0</td>
</tr>
<tr>
<td>2011</td>
<td>18.6±0.4</td>
<td>17.4±0.7</td>
<td>18.1±0.5</td>
<td>17.6±0.8</td>
</tr>
<tr>
<td>N$_6$ 2007</td>
<td>5.9±0.9</td>
<td>9.4±2.0</td>
<td>9.6±2.1</td>
<td>10.5±1.9</td>
</tr>
<tr>
<td>2011</td>
<td>7.9±1.3</td>
<td>12.2±1.9</td>
<td>12.9±1.6</td>
<td>19.2±2.5</td>
</tr>
<tr>
<td>NO$_3^-$ 2007</td>
<td>5.6±0.8</td>
<td>8.3±1.9</td>
<td>8.9±2.2</td>
<td>8.1±1.7</td>
</tr>
<tr>
<td>2011</td>
<td>7.3±1.3</td>
<td>9.8±1.1</td>
<td>12.0±1.5</td>
<td>17.4±2.3</td>
</tr>
<tr>
<td>NH$_4^+$ 2007</td>
<td>0.3±0.1</td>
<td>1.1±0.3</td>
<td>0.7±0.1</td>
<td>2.4±0.3</td>
</tr>
<tr>
<td>2011</td>
<td>0.6±0.1</td>
<td>2.3±1.0</td>
<td>0.9±0.1</td>
<td>1.8±0.5</td>
</tr>
<tr>
<td>OM (%) 2007</td>
<td>5.7±0.7</td>
<td>5.8±0.3</td>
<td>6.4±0.3</td>
<td>7.0±1.2</td>
</tr>
<tr>
<td>2011</td>
<td>4.9±0.5</td>
<td>6.5±0.3</td>
<td>7.0±0.5</td>
<td>7.8±0.5</td>
</tr>
<tr>
<td>pH(H$_2$O) 2007</td>
<td>5.8±0.2</td>
<td>6.5±0.3</td>
<td>6.4±0.1</td>
<td>6.6±0.4</td>
</tr>
<tr>
<td>2011</td>
<td>5.1±0.2</td>
<td>4.9±0.3</td>
<td>5.1±0.1</td>
<td>5.6±0.3</td>
</tr>
</tbody>
</table>

Soil surface (0–15 cm) properties [total N and C, C/N ratio, concentrations of N$_6$, extractable inorganic N, NO$_3^-$ – nitrate, and NH$_4^+$ – ammonium, OM – organic matter, and pH (H$_2$O)], in the first (May 2007) and fifth (May 2011) springs of the experiment, according to the N addition treatment (Control, 40A, 40AN and 80AN). Different letters refer to significant differences between treatments (ANOVA $p<0.05$ followed by a Bonferroni test), while * refers to significant differences between 2007 and 2011 (there were no significant interactions between time and treatment, $p>0.05$ – Table S3). Values represent the mean ($n=3$ experimental plots per treatment) ± SE.

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Are there consistent plant responders to increased N availability in Mediterranean maquis?

The composition of the plant community found at the experimental site was consistent with that expected in a Mediterranean maquis regenerating after a fire event [46]. After a fire, there is a ‘window of opportunity’, *i.e.*, a period of reduced competition for light, nutrients and water [40,41], which favours the establishment of many annuals, hemicryptophytes, geophytes and subshrubs as well as ruderal species colonizing from the surroundings [46]. In a post-fire succession, as the ‘window of
opportunity’ passes (~5 years after the fire), some plant species tend to disappear from the community [47]. This natural dynamic of the plant community [22,46] has to be taken into account as the background over which the population dynamics associated with the N additions have to be superimposed.

After one year, D. viscosa was the only species which had benefited from N addition (irrespective of dose and form). In subsequent years, its cover, together with that of Sanchus sp., began to decrease, which is consistent with their disappearance in late succession [40,41,48]. However, N additions appeared to offset some of the cover loss observed in the control, suggesting that increasing N availability extends the ‘window of opportunity’ for more nutrient-demanding plants. Other plant species responded more consistently to the N dose and/or form over the 5-year study period suggesting that they could be used as indicators of the N status of the ecosystem for at least a period of 5 years.

Are the N-driven changes in diversity linked with functional alterations?

N-addition treatments failed to change soil C and N concentrations, or the C/N ratio, which remained within the range reported for the Mediterranean Basin [49]. In contrast to most studies [6], soil acidity was unaffected by the N additions, possibly due to the high soil calcium carbonate content [50]. However, after 5 years, application of the highest N dose (80AN) had increased soil extractable N, possibly indicating the transition from a closed N cycle to an open and leaky one where N may be lost through NO₃⁻ leaching [51]. However, NO₃⁻ leaching can occur much later than many diversity and internal N cycle changes [8]. On the other hand, and in contrast to observations of several studies [52], high N (80AN) addition does appear to have increased soil organic matter, most likely reflecting a decrease in decomposition [53]. This is of particular importance for Mediterranean Basin soils due to their naturally low organic matter concentration and hence higher susceptibility to erosion and desertification [54]. Altogether, it can be concluded that the

Figure 4. Impacts of the N treatments on plant biomass. Aboveground plant biomass (a), estimated litterfall production per year (b) and total aboveground biomass (sum of the standing biomass and the cumulative litterfall produced on the previous four years - c) according to the N treatments (Control, 40A, 40AN and 80AN). Aboveground plant biomass was harvested in June 2011 (the fifth spring of the experiment) from three 1-m² squares per experimental plot. Litterfall was collected from April to December 2010 using litter traps (five 0.04 m² per experimental plot). Different letters refer to statistically significant differences between treatments (ANOVA p<0.05 followed by a Bonferroni test). Bars represent the mean (n = 3 experimental plots per treatment) ±SE.

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cumulative N inputs have increased the ‘N status’ of the ecosystem [7,55].

The aboveground biomass accumulated under control conditions was within the range of the equivalent Californian habitat – coastal sage scrub [36]. Under increased N availability, and in agreement with reports of other temperate [6] and Mediterranean ecosystems [18,30,56], aboveground plant biomass increased. Positive relationships between plant richness and productivity have been reported for Mediterranean [57] and several other ecosystems [58,59]. However, in this study, plant richness and plant productivity were not related, as these were promoted by different N sources. The treatments with more NH4+-N (40A and 80AN) promoted species richness, but the incomers were small plants that contributed very little to productivity or to protect the soil from erosion that may result from the increased % of bare soil. Given that the non-random N-driven loss of the dominant plant species has been shown to reduce productivity [3], the NH4+-driven reduction in the cover of the most abundant plant species (e.g., Cistus ladanifer, Ulex densus) would explain the absence of a productivity increase.

In general, N-enhanced plant richness was not linked with functional alterations, mainly because the N-benefited species were small short-lived plants, providing a minor contribution to ecosystem functions while the N-affected plants were perennial shrubs, which underpin ecosystem functions.

Is the form of N more important than the dose?

The data suggest that plant community composition is driven by the amount of NH4+-N, highlighting the importance of NH4+-N as a driving force in Mediterranean ecosystems [17,26]. Given the characteristics of the Mediterranean climate (long dry periods interspersed with torrential rain events), which facilitates both rainfall and wind erosion [60], plant cover and the spatial structure of vegetation play a significant role in preventing soil loss and erosion in Mediterranean areas [61]. Due to the likely loss of cover with increasing NH4+ deposition, agricultural emissions are likely to increase erosion in the maquis. This suggests that enhanced N availability (resulting mainly from agriculture) may affect the structure of the surrounding maquis, decrease soil protection (as a consequence of an increase in bare soil) and promote N leakage (as a consequence of an increase in soil extractable N).

In contrast, the lower NH4+-N dose applied to the combined N treatment (40AN) plots, with fewer plant species, improved aboveground biomass and soil protection most effectively by almost fully covering the soil with plants. Erosion can also be increased indirectly as a result of NO3 deposition increasing aboveground biomass and the risk of wildfires, indicating that increasing urbanisation also threatens these ecosystems. Land managers and planners need to be aware of these effects before they authorise expansion of such N sources close to these ecosystems.

Overall, these results highlight the importance of cumulative N and enhanced N availability, particularly of NH4+-N, as a driving force behind the dynamics and stability in plant community structure and composition of Mediterranean ecosystems [17,26]. The high resilience to disturbance of plant communities in the Mediterranean Basin has been related to their evolutionary history of disturbance, namely anthropogenic pressure [62]. Also, biodiversity may have its greatest impact on the functioning of diverse, naturally heterogeneous ecosystems [63], so it is possible that patchiness and heterogeneity may confer higher resilience to increased N availability (and hence higher reliability on ecosystem functions) than homogeneous un-fragmented ecosystems. Finally, the effectiveness of the critical N loads [64] established for this European habitat, at between 20 and 30 kg N ha–1 yr–1 [65] would be improved by inclusion of the impact of the N form.

Supporting Information

Table S1 Effect of the N treatments on plant community composition and cover. List of the vascular plant species observed in the three assessments, and their respective changes in cover (2008–2007/2011–2007) according to the N additions. (DOCX)

Table S2 List of the plant species that responded consistently (after one and 5 years of N addition treatments) to the N dose and/or form. The species’ most common habitats and, when available, their responses to N enrichment in other studies are shown in the right column. (DOCX)

Table S3 Statistical analyses of soil surface properties. (DOCX)

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Author Contributions

Conceived and designed the experiments: TD MAML LS CC. Performed the experiments: TD AC. Analyzed the data: TD AC MAML LS RB CC. Contributed reagents/materials/analysis tools: TD AC MAML CC. Wrote the paper: TD AC MAML LS RB CC. Obtained permission for use of field site: TD CC.

Figure 5. Impact of the N treatments on the % of bare soil.

Response of the % of bare soil to the N treatments (Control, 40A, 40AN and 80AN) on the first and fifth springs of N additions (2007 and 2011). Different letters refer to statistically significant differences between treatments (two-way ANOVA p<0.1 followed by a Bonferroni test; there were no significant interactions between treatment and time). Bars represent the mean (n = 3 experimental plots per treatment) ±SE. doi:10.1371/journal.pone.0092517.g005

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