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Economic Perspective on Grasslands, Biodiversity and Weather Extremes

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Abstract

Grasslands cover large shares of the world's terrestrial area and are essential for the global food security as well as for the agricultural economy as they provide feed for animals. The same grasslands are a host of biodiversity and provide a wide range of other ecosystem services. Changes in the climatic conditions, population, consumer demands and policies, which put pressure on grasslands and grassland-based production, requires farmers to adapt. For farmers and their utility, two aspects are especially important in this context: expected income and stability of income, i.e. income risk.

Farmers have a set of tools to adapt and influence grassland production, including the choice of the (plant) species diversity level in their grassland. Species diversity have been shown to increase yields and yield stability as well as that it can mitigate adverse effects of weather extremes, such as droughts. A key challenge is to transform these findings also into economic terms and to quantify these potential benefits of species diversity in terms of farmers' utility. Empirical economic assessments of the effects of species diversity ('diversity effects') in grasslands that aim to value these effects from farmers' perspectives have been conducted only in a limited number. Those studies that did such economic assessment usually focused on biomass yields (i.e. quantity of yield) and extensively managed grasslands and neglected essential aspects for farmers, such as forage quality, different management intensities, extreme weather events (e.g. droughts), market responses (e.g. hay prices) or differences in farm types.

Droughts that are a severe risk for grasslands are predicted to increase in frequency and intensity in the future due to climate change. Understanding the economic consequences of drought effects and how species diversity, among other tools, can help to reduce adverse effects is important. Moreover, droughts can not only cause yield losses but also market response (e.g. hay prices response), which are equally important to farmers but not explored so far.

The overall research goal of the thesis is to support farmers, extension services and policy makers by answering the question '*What is the value of species diversity in grasslands for farmers?*'. Therefore, we conduct an economic valuation of the diversity effect in grasslands from a farmers' perspective in *Chapter 2 to 4*: In *Chapter 2*, we economically assess the diversity effects on expected yields, expected revenues and risk in intensively managed grasslands, considering biomass yields and forage quality. We find that species diversity increased biomass yields while it did not affect forage quality. Thus, species diversity increased quality-adjusted yields (biomass yields x forage quality) and potential revenues from milk production. Moreover, species diversity decreased farmers' risk. In *Chapter 3*, we investigate the diversity effect on expected yields and revenues, considering biomass yields and forage quality across a wide range of management intensities. The results show that species diversity increased quality-adjusted yields and revenues across management regimes and that the benefits from species diversity can be as high as increasing management intensity. In *Chapter 4*, we analyze the

diversity effect under increasing drought risks while considering hay yield and price responses to droughts as well as different farm types. We find that species diversity increased income and decreased risk. The diversity effect was independent of drought probability and the drought effect largely dependent on farm type.

We complement the economic assessment of species diversity by an in-depth analysis of droughts effect hay prices and differences between drought effects on hay prices and feed grain prices (*Chapter 5*). We find that regional and national droughts substantially increased hay prices while feed grain prices were not affected in South Germany. The differences between hay and feed grain price responses can be linked to transport and transaction costs, thus, to market integration.

Moreover, we provide supplementary analysis about biodiversity loss communication, public interest in biodiversity in climate change and societal concerns on pesticide use in the appendices of the thesis. In conclusion, the thesis adds to the existing knowledge of species diversity and its economic value by extending the (economic) assessment of species diversity. Considering key aspects for farmers, we show that species diversity is an economically relevant factor of production and that maintaining and reestablishing species diverse grasslands can contribute to future sustainable intensification of grassland-based production. Moreover, we provide new insights to the consequences of droughts on hay and feed grain prices and that these prices are differently affected. The knowledge on these drought effects on prices in combination with diversity effects in grasslands can help to better understand and mitigate adverse drought effects in grassland-based production, especially considering climate change. The gained insights in this thesis, about the species diversity effects and the drought effects, should be considered in farm management as well as in the decisions of policy makers.

Zusammenfassung

Grasland bedeckt einen großen Teil der globalen Landfläche und ist sowohl für die weltweite Ernährungssicherheit als auch für die Agrarwirtschaft von wesentlicher Bedeutung, da es als Futtermittelquelle für Tiere dient. Dasselbe Grasland beherbergt zudem eine Vielzahl von Arten und bietet zahlreiche weitere Ökosystemdienstleistungen. Veränderungen der klimatischen Bedingungen, der Bevölkerungszahl, der Verbraucherwünsche und der Politik – welche Druck auf das Grasland und die graslandbasierte Produktion von Lebensmitteln ausüben – erfordern Anpassungen der Landwirtinnen und Landwirte. Für die Landwirtinnen und Landwirte und ihren ökonomischen Nutzen sind in diesem Zusammenhang zwei Aspekte besonders relevant: das erwartete Einkommen und die Einkommensstabilität, d.h. das Einkommensrisiko.

Die Landwirtinnen und Landwirte verfügen über eine Reihe von Instrumenten zur Anpassung ihres Graslands, einschließlich der Wahl der (Pflanzen-)Artenvielfalt. Forschungsergebnisse haben gezeigt, dass die Artenvielfalt die Erträge und die Stabilität des Graslands erhöht sowie dass die Artenvielfalt negative Effekte von Wetterextremen, wie Dürren, reduzieren kann. Eine zentrale Herausforderung besteht darin, diese Erkenntnisse und die potenziellen Vorteile der Artenvielfalt im Hinblick auf den ökonomischen Nutzen für die Landwirtinnen und Landwirte zu quantifizieren. Empirische ökonomische Bewertungen der Effekte der Artenvielfalt ("Diversitätseffekte") im Grasland aus einer landwirtschaftlichen Sicht wurden bisher nur in einer begrenzten Anzahl durchgeführt. Die Studien, die eine solche ökonomische Bewertung vornahmen, konzentrierten sich in der Regel auf Biomasserträge (d.h. die Ertragsmenge) und extensiv bewirtschaftetes Grasland und vernachlässigten wesentliche Aspekte für die Landwirtinnen und Landwirte, wie z.B. Futterqualität, unterschiedliche Bewirtschaftungsintensitäten, extreme Wetterereignisse (z.B. Dürren), Marktreaktionen (z.B. Heupreise) oder Unterschiede in den Betriebstypen.

Dürren, die ein großes Risiko für Grasland darstellen, werden aufgrund des Klimawandels in Zukunft voraussichtlich häufiger und intensiver auftreten. Es ist wichtig, die wirtschaftlichen Folgen von Dürreeffekten zu verstehen und zu verstehen, wie die Artenvielfalt, neben anderen Instrumenten, dazu beitragen kann, die negativen Auswirkungen zu reduzieren. Darüber hinaus können Dürren nicht nur Ertragseinbußen verursachen, sondern auch Marktreaktionen (z.B. Heupreisereaktionen) auslösen. Diese Marktreaktionen sind für Landwirtinnen und Landwirte ebenso wichtig wie Ertragseinbußen, sind aber bisher noch nicht genauer untersucht worden.

Das übergeordnete Forschungsziel dieser Arbeit ist die Unterstützung von Landwirtinnen und Landwirte, Beratungsdiensten und politischen Entscheidungsträgern durch die Beantwortung der Frage 'Was ist der Mehrwert von Artenvielfalt für Landwirtinnen und Landwirte im Grasland?'. Daher führen wir in *Kapitel 2 bis 4* eine ökonomische Bewertung der Diversitätseffekte im Grasland aus landwirtschaftlicher Perspektive durch: In *Kapitel 2* bewerten wir ökonomisch die Diversitätseffekte

auf die erwarteten Erträge, die erwarteten Einnahmen und das Risiko in intensiv bewirtschaftetem Grasland. Dabei berücksichtigen wir Biomasseerträge als auch die Futterqualität. Wir finden, dass Artenvielfalt die Biomasseerträge erhöht, während sie die Futterqualität nicht beeinflusst. Die Artenvielfalt erhöhte somit die qualitätskorrigierten Erträge (Biomasseerträge x Futterqualität) und die potenziellen Einnahmen aus der Milchproduktion. Darüber hinaus verringerte die Artenvielfalt das Risiko der Landwirtinnen und Landwirte. In *Kapitel 3* untersuchen wir den Diversitätseffekt auf die zu erwartenden Erträge und Einnahmen. Dabei berücksichtigen wir Biomasseerträge und die Futterqualität über ein weites Spektrum an Bewirtschaftungsintensitäten. Unsere Resultate zeigen, dass Artenvielfalt die qualitätskorrigierten Erträge und Einnahmen über die verschiedenen Bewirtschaftungsregimes hinweg erhöht hat und dass die Vorteile der Artenvielfalt ebenso hoch sein können wie die Erhöhung der Bewirtschaftungsintensität. In *Kapitel 4* analysieren wir den Diversitätseffekt unter zunehmendem Dürreerisiko. Dabei berücksichtigen wir den Einfluss von Dürre auf Heupreise und verschiedene Betriebstypen. Wir finden, dass die Artenvielfalt das Einkommen erhöht und das Risiko verringert. Der Diversitätseffekt war unabhängig von der Dürrewahrscheinlichkeit und der Dürreeffekt auf Einkommen und Risiko war weitgehend vom Betriebstyp abhängig.

Ergänzend zur ökonomischen Bewertung der Artenvielfalt analysieren wir die Auswirkungen von Dürren auf die Heupreise und Unterschiede des Dürreeffekts auf Heupreise und Futtergetreidepreise (*Kapitel 5*). Wir finden, dass regionale und nationale Dürren die Heupreise in Süddeutschland erheblich erhöhten, während die Futtergetreidepreise nicht betroffen waren. Die unterschiedlichen Reaktionen von Heu- und Futtergetreidepreisen können mit den Transport- und Transaktionskosten, und damit mit der Marktintegration, in Verbindung gebracht werden.

Zudem bieten wir ergänzende Analysen in den Anhängen dieser Arbeit zu den Themen Kommunikation des Biodiversitätsverlusts, Interesse der Öffentlichkeit in den Themen Klimawandel und Biodiversität und gesellschaftlichen Bedenken in Bezug auf Pflanzenschutzmitteleinsatzes.

Zusammenfassend ergänzt diese Arbeit durch Erweiterung der (ökonomischen) Bewertung von Artenvielfalt das vorhandene Wissen über Artenvielfalt und dessen ökonomischen Mehrwert. Wir zeigen, dass Artenvielfalt ein ökonomisch relevanter Produktionsfaktor ist und dass die Erhaltung und Wiederherstellung artenreicher Graslandschaften zu einer nachhaltigen Intensivierung der graslandbasierten Produktion beitragen kann. Darüber hinaus liefern wir neue Erkenntnisse über die Auswirkungen von Dürren auf Heu- und Futtergetreidepreise und dass diese Preise unterschiedlich durch Dürre beeinflusst werden. Die Erkenntnisse über diese Dürreeffekte auf Preise zusammen mit den Diversitätseffekte im Grasland kann dazu beitragen, die negativen Auswirkungen von Dürren auf die graslandbasierte Produktion zu verstehen und abzuschwächen. Dies ist besonders wichtig unter Berücksichtigung des Klimawandels. Die in dieser Arbeit gewonnenen Erkenntnisse über die Effekte

von Artenvielfalt und von Dürren sollten sowohl bei der landwirtschaftliche Betriebsführung als auch bei Entscheidungen von politischen Entscheidungsträger berücksichtigt werden.

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1 Chapter 1 – Introduction

Grasslands cover large shares of the world's terrestrial area and are essential for the global food security as well as for the agricultural economy as they provide feed for animals (Sanderson et al. 2004, Soussana and Lüscher 2007, Huyghe et al. 2014). The same grasslands are a host of biodiversity and provide a wide range of other ecosystem services (Le Clec'h et al. 2019). Changes in the climatic conditions, biodiversity loss, population, consumer demands and policies, which put pressure on grasslands and grassland-based production, requires farmers to adapt. For farmers and their utility, two aspects are especially important in this context: expected income and stability of income, i.e. income risk.

Farmers, have several ways to manage their grasslands to influence yields and production risks, such as cutting frequency or irrigation. Moreover, grassland (plant) species diversity, and farmers decision about it, influences grasslands. Species diversity have been shown to increase yields and yield stability as well as that it can mitigate adverse effects of weather extremes, such as droughts (Tilman et al. 1996, Isbell et al. 2009, Marquard et al. 2009, Van Ruijven and Berendse 2010, Finn et al. 2013, Isbell et al. 2015, Haughey et al. 2018). A key challenge is to transform these findings also into economic terms. Current economic assessments of the effects of species diversity ('diversity effects') in grasslands are limited. Those studies that did such economic assessment usually focused on biomass yields (i.e. quantity of yield) and extensively managed grasslands and neglected essential aspects for farmers, such as forage quality, different management intensities, extreme weather events (e.g. droughts), market responses (e.g. hay prices) or differences in farm types.

Droughts are a severe risk for grasslands and are predicted to increase in frequency and intensity due to climate change (Ciais et al. 2005, Smit et al. 2008, IPCC 2013, Spinoni et al. 2018). Understanding the economic consequences of drought effects and how species diversity, among other tools, can help to reduce adverse drought effects is important. Moreover, droughts can not only cause yield losses but also market response (e.g. hay prices response), which are equally important to farmers but not explored so far. The drought responses of hay prices also might differ from other (feed) price drought responses.

In the remainder of the thesis, we first provide a brief overview of the effect of species diversity in grasslands and its economic assessment as well as of drought effects on hay prices and important mechanisms in this context (1.1). This is followed by an outline of the research questions and of the thesis structure (1.2) as well as by a presentation of the thesis contributions (1.3). In the next section, we conclude the main thesis results and provide an outlook for future research (1.4). Finally, we present the main original research articles in *Chapter 2 to 5* and supplementary original research articles in *Thesis Appendix 1 to 3*.

1.1 Background

Farmers can influence the level of species diversity when managing their grasslands (Fig. 1). For example, farmers can do this by sowing or oversowing grasslands with mixtures or adjusting fertilization and cutting regimes (Walker et al. 2004). Species diversity can create benefits for farmers (through the provision of ecosystem services). Thus, species diversity can increase farmers' utility. Species diversity can increase farmers' utility by increasing the expected yield, thus income, and reducing production risk, thus income risk (Baumgärtner 2007). The latter depends on farmers risk preferences (i.e. if the farmer is risk averse, risk neutral or risk loving) and how strong these risk preferences are (Chavas 2004). How species diversity affects grasslands, thus, farmers' utility also depends on other factors (e.g. management intensity or droughts). Therefore, species diversity and these factors are jointly influencing farmers' utility. Next to this, additional other factors exist that are not directly interacting with species diversity (in the way how species diversity affects grasslands) but also influence grasslands, farmers' utility and farmers decisions (e.g. hay prices or policies). In a reciprocal process farmers' decision, factors that affect grasslands and benefits retrieve from grasslands influence each other.

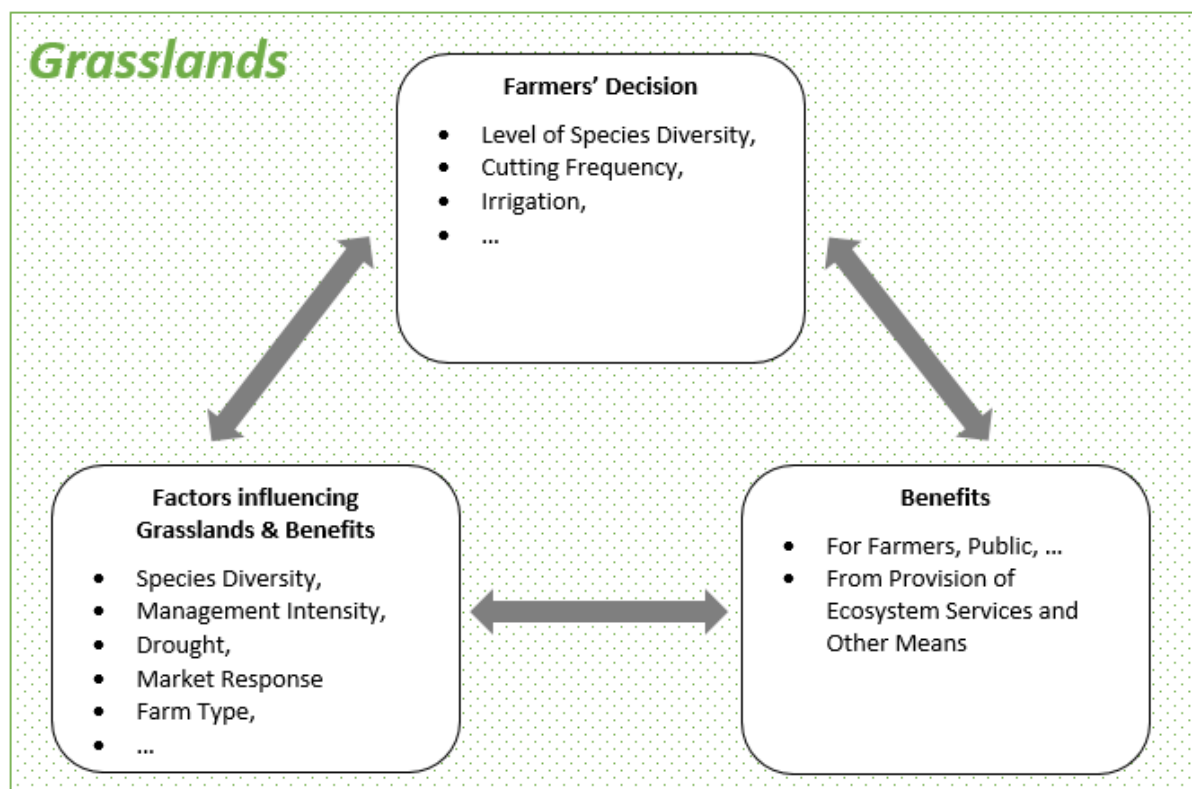


Figure 1: Overview of farmers, diversity effects and its interdependences in grasslands. The illustration focuses on topics covered in within this thesis and is not supposed to be an exhaustive depiction.

1.1.1 Diversity Effects in Grasslands and its Economic Assessment

The specific ways how species diversity can affect grasslands, which are especially important from a farmers' perspective, are: Species diversity can affect biomass yields, forage quality and quality adjusted yields (biomass yields x forage quality). Previous research showed a positive diversity effects on biomass yields (Tilman et al. 1996, Marquard et al. 2009, Finn et al. 2013), which were also shown across different management intensities (Weigelt et al. 2009, Craven et al. 2016). In contrast, findings on the diversity effects on forage quality are ambiguous. However, in experiments these effects are often found to be small (Deak et al. 2007, Sturludóttir et al. 2014, Oelmann et al. 2015, Ergon et al. 2017). The diversity effects on quality-adjusted yields were as a result often reported to be positive (Nyfeler et al. 2011, Oelmann et al. 2015, Ergon et al. 2017). These diversity effects on quality-adjusted yields were usually driven by strong diversity effects on biomass yields. Whether these effects hold across management intensities, a wide species diversity gradient and different forage quality variables is not answered. The positive diversity effect on biomass yields has been linked to the complementarity and sampling effects (Tilman et al. 1997, Loreau and Hector 2001; but see Barry et al. (2019)). The complementarity effect is assumed to be caused by resource partitioning or positive interaction of species, while the sampling is based on a higher probability of including highly productive species in more species diverse grasslands.

Furthermore, species diversity is hypothesized to reduce negative effects of perturbations, such as droughts, on yields and yield stability. In the ecological literature this is referred to as the 'insurance effect' of species diversity (Yachi and Loreau 1999). Previous studies showed that species diversity can increase temporal stability of biomass yields and quality-adjusted yields (Isbell et al. 2009, Haughey et al. 2018, Schaub et al. 2020). Results of whether species diversity increase resistance (i.e. perturbation induced yield reduction) and resilience (i.e. return to pre-perturbation yield; sensu Van Ruijven and Berendse (2010)) against weather extremes remain however ambiguous (Van Ruijven and Berendse 2010, Vogel et al. 2012, Isbell et al. 2015, Wright et al. 2015, Craven et al. 2016).

A key challenge is to transform these findings also into economic terms and to quantify these potential benefits of species diversity in terms of farmers' utility. Empirical economic assessments of the effects of species diversity in grasslands that aim to value these effects from farmers' perspectives have been conducted only in a limited number (Schlöpfer et al. 2002, Koellner & Schmitz 2006, Dörschner & Musshoff 2013, Finger & Buchmann 2015, Binder et al. 2018). The results show that species diversity increase expected revenues and that the risk-reducing effect of species diversity on yields creates significant insurance values for risk-averse farmers. This has important implications. First, species diversity can sustainably increase farm revenues. Second, species diversity can serve as valuable ex-ante risk management strategy for farmers. Third, farmers' risk preferences may shape agroecosystem

diversity. Fourth, not accounting for the yield stabilizing property of species diversity would lead to an underestimation of its economic value.

However, there are several shortcomings in this literature. First, current assessments focus on diversity effects on biomass yields. However, equally important to farmers are forage quality and quality-adjusted yields. Quality-adjusted yields is an integrated measure of biomass yields and forage quality, which describes how much quality, for example energy, is available per area. One exception from the focus on biomass yields in the economic assessment of species diversity is Binder et al. (2018). Binder et al. (2018) considered next to biomass yields also protein content in extensively managed grasslands. Second, economic literature on species diversity effect have not accounted for interactions of the diversity effects and management intensity; although such assessments could help decision makers in comparing the economic benefits along the increasing species diversity and management intensity. Third, extreme weather events, such as droughts, are a severe risk for grasslands (Ciais et al. 2005, Smit et al. 2008). These effects and the insurance value of species diversity in such events have potential economic value, however, they have not been assessed so far. Fourth, current economic assessments have not accounted for the interdependencies among species diversity, differences in farm types and the market response (e.g. hay price) to shocks.

1.1.2 Hay Prices & Droughts

For farmers prices (e.g. hay prices) are equally important as yields (e.g. hay yields). For example in the event of droughts price increases can balance drought induced yield losses, thus, stabilize farmers' income. This balancing is connected to the natural hedge, i.e. a negative correlation of prices and yields (McKinnon 1967). How prices react to droughts depends, amongst others, on the product's transport and transaction costs, thus, potential market integration (Goodwin and Piggott 2001; Balcombe et al. 2007). Market integration is linked to how one market (or region) can balance yield losses of another market. The price response (as well as yield response) to droughts have different implication depending on the farm type, i.e. whether farmers are net sellers or buyers. Understanding hay price dynamics in the event of a drought and how these are different from other agricultural products is important, however, so far not empirically studied. Moreover, when economically assessing the species diversity as a tool to mitigate drought effects it is important to consider drought induced price responses.

1.2 Research Goal & Thesis Structure

The overall research goal of the thesis is to answer: '*What is the value of species diversity in grasslands for farmers?*'. In answering this question we aim to support farmers (in choosing the optimal level of species diversity), extension services as well as policy makers. In our assessment we follow an

interdisciplinary approach (e.g. by utilizing both economic and ecological theories and data from the ecological, economic and meteorological communities). We contribute with our economic assessment of species diversity to close the gaps outlined in the previous section. In detail, we answer the following research questions (Fig. 2):

- *Research questions 1:* What is the value of species diversity for farmers when considering quantity and quality of yields and risk in intensively managed grasslands? (*Chapter 2*, see yellow area in Fig. 2)
- *Research questions 2:* What is the value of species diversity for farmers when considering quantity and quality of yields across a wide range of management intensities in grasslands? (*Chapter 3*, see blue area in Fig. 2)
- *Research questions 3:* What is the value of species diversity under increasing drought risks? Focusing on expected value and risk reduction as well as differences between farm types (hay farmers vs. milk farmers; *Chapter 4*, see red area in Fig. 2)

Furthermore, we complement the economic assessment by in-depth investigating the effects of droughts on hay prices and how these drought effects differ between hay and other feed price.

- *Research questions 4:* What is the effect of droughts on hay and feed grain prices? (*Chapter 5*, see area surrounded by dashed lines in Fig. 2)

Finally, in the appendices of the thesis we additionally asked the questions related to the topic of the thesis:

- *Supplementary research questions 1:* Why are we experiencing such unrestrained biodiversity loss and how can we improve communication to reduce (our tolerance towards) biodiversity loss? (*Thesis Appendix 1*, see grey area in Fig. 2)
- *Supplementary research questions 2:* What are the global trends in public interest in climate change and biodiversity and how are these interest trends affected by research-policy efforts and activism? (*Thesis Appendix 2*, see grey area in Fig. 2)
- *Supplementary research questions 3:* What is the public interest in pesticides over time in Switzerland and how does it relate to popular initiatives? (*Thesis Appendix 3*, see grey area in Fig. 2)

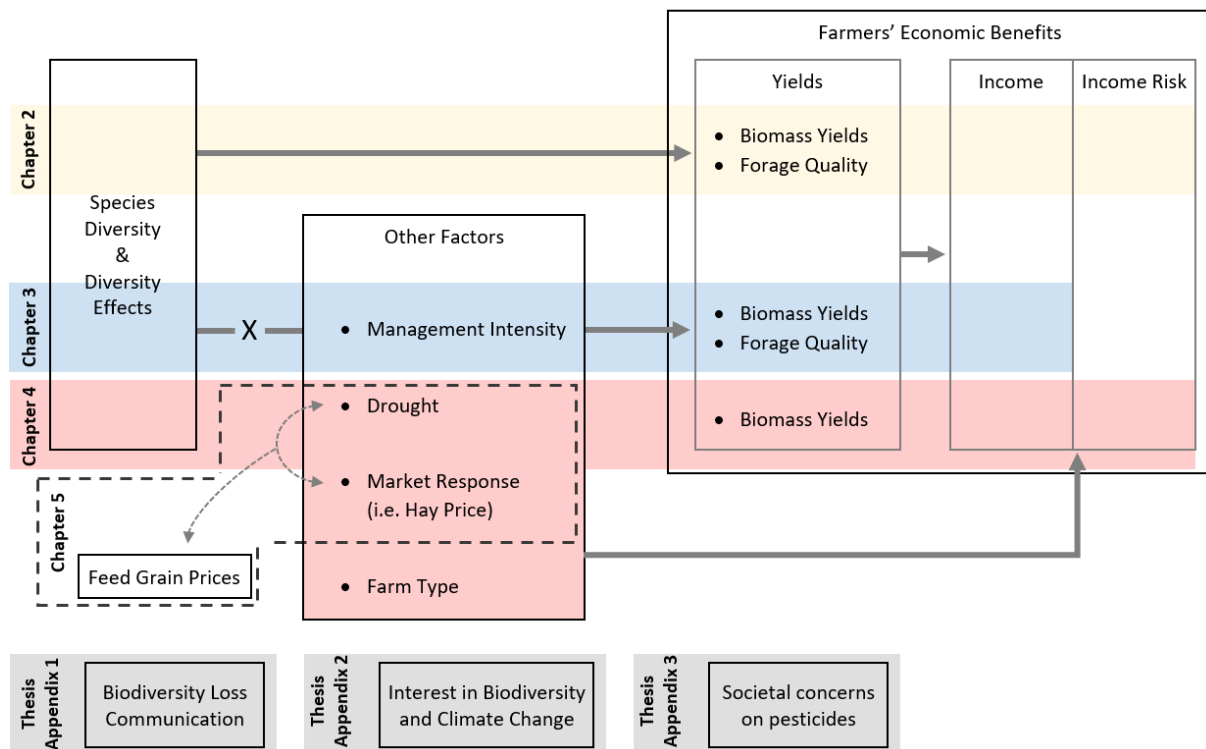


Figure 2: Thesis outline and diversity effects and its interdependences in grasslands. See Fig. 1 for additional details. *Chapter 2* and *3* focus on revenues while *Chapter 4* on income, therefore, the term income in *Chapter 1* can refer either to income or revenues.

1.3 Thesis Contributions

In this section we summarize the main findings of the research articles (*Chapter 2 to 5*). We start with the research question 1 to 3. These are dealing directly with the ‘*economic valuation of the diversity effects in grasslands*’ (*Chapter 2 to 4*). This is followed by the complementary research question 4, which deals with the ‘*drought effect on hay and other feed prices*’ (*Chapter 5*). The details about the supplementary research question 1 to 3 are only presented in the *Thesis Appendix 1 to 3*.

1.3.1 Economic valuation of the diversity effects in grasslands

In *Chapter 2*, we economically assess the diversity effects on expected yields, expected revenues and risk in intensively managed grasslands, considering biomass yields and forage quality. We find that species diversity increased biomass yields while it did not affect forage quality. Thus, species diversity increased quality-adjusted yields and potential revenues from milk production. Moreover, species diversity decreased farmers’ risk. The combined economic species diversity value of expected revenue increase and risk reduction for farmers, expressed in certainty equivalent¹ (see e.g. Chavas 2004), was

¹ The certainty equivalent is a certain payment that provides farmers with the same utility as a stochastic payment (see e.g. Chavas 2004). Note that we assume farmers to be rather risk averse (see e.g. Hardaker et al 2015) if not mentioned differently.

about +1630 Euro ha⁻¹ a⁻¹ (+29%) when comparing the average mixture to the average monoculture. For risk averse decision makers, this gain in certainty equivalent was mainly due to the increase in revenues (accounting for 90%) compared to the value gain from risk reduction (accounting for 10%). We show in this chapter that farmers benefit economically from species diversity and that even a moderate increase in this diversity contributes to more stable grassland-based production.

In *Chapter 3*, we investigate the diversity effect on expected yields and revenues, considering biomass yields and forage quality across a wide range of management intensities. The results show that species diversity increased quality-adjusted yields and revenues across management regimes. On average of all management intensities, an increase of species diversity from 1 to 16 (1 to 60) species increased revenues by about +1400 Euro ha⁻¹ a⁻¹ (+3100 Euro ha⁻¹ a⁻¹) in semi-natural grassland. Moreover, species diversity was found to be as valuable as increasing the fertilization level and cutting frequency. Based on this chapter, we conclude that species diversity can substantially and sustainably increase quality-adjusted yields and revenues in semi-natural grasslands.

In *Chapter 4*, we analyze the diversity effects under increasing drought risks, i.e. increasing drought probability, while considering hay yield and price responses to droughts as well as different farm types. We show that species diversity increased income and decreased risk. These effects were independent of drought probability. For net farmers (i.e. farmers without own hay demand) the certainty equivalent was about +4302 Euro farm⁻¹ a⁻¹ (+25%) higher when comparing the average mixture to the average monoculture. For net buyers (i.e. farmers with 130% hay demand of their own production) the equivalent analysis resulted in gains of +4258 Euro farm⁻¹ a⁻¹ (+8%). Furthermore, higher drought probabilities did not affect net sellers while net buyers certainty equivalent decreased by -684 Euro farm⁻¹ a⁻¹ (-1%). Consequently, we show in this chapter that differences in farm types need to be addressed when assessing drought effects and that species diversity is also an important production factor when considering increasing drought risks.

1.3.2 Drought effect on hay and other feed prices

In *Chapter 5*, we analyze how regional (i.e. in South Germany) and national (i.e. in whole Germany) droughts affected hay and feed grain prices in South Germany. We find that regional and national droughts substantially increased hay prices by up to 15%. The prices responded with a delay of about three months and the drought induced price response lasted for about year after a drought event. In contrast, feed grain prices in our sample were neither affected by regional nor by national droughts. The differences between hay and feed grain price responses can be linked to market integration, as hay markets are usually regionally organized while feed grains are traded transnationally. It is important to include these insights into farm management and policy actions, especially considering predicted increases in drought frequency and intensity under climate change. Here, increasing market

transparency by, for example, online feed price exchanges might contribute to reduce price shocks as it increases potential market integration.

1.4 Thesis Conclusions & Research Outlook

The thesis adds to the existing knowledge of diversity effects in grasslands and its economic valuation as well as to understand drought effects on hay and feed grain prices. In the analyses of the thesis, we include several new aspects to the economic species diversity assessment that are crucial for farmers, including forage quality, quality-adjusted yields, risk, a range of management intensities, increasing drought risks, market response and differences in farm types. Overall, we conclude that species diversity is an economically relevant factor of production. Maintaining and reestablishing species diverse grasslands can provide a win-win situation as it contributes to future sustainable intensification of grassland-based production, while at the same time it can support important ecosystem services and functions. Moreover, the knowledge of drought effects on hay prices (and how they differ from drought effects on feed grain prices) in combination with diversity effects can help to better understand and mitigate adverse drought effects in grassland-based production, especially considering climate change. The gained insights in this thesis, about the species diversity effects and the drought effects, should be considered in farm management as well as in the decisions of policy makers.

The economic assessment of species diversity presented in this thesis offers several entry points and extensions for future work. For example: i) Analyzing the differences between the social and private optimal species diversity level given market price responses, risk aversion and differences in farm types. This can be extended by also considering how policies and insurances impact these optima and if policies and insurances can be used to align the social and private optimum. ii) An assessment of the costs for maintaining and reestablishing species diverse grasslands (e.g. assessing the relationship between the prices of seed mixtures and its species diversity), iii) Replication and extension of the economic assessments conducted in the thesis with new data² and/or different weather extremes (e.g. extreme temperatures or floods), iv) Extension of the economic species diversity assessment to other land covers, such as croplands or forest, or to other trophic levels than plant species (see e.g. Di Falco and Chavas 2006 or Kremen 2018).

In the analysis of drought effects on hay and feed grain prices, we explore drought effects based on low precipitation and high temperature events, which together increase intensity of droughts and often occur together (Trenberth and Shea 2005; Estrella and Menzel 2013). Additionally to these weather events, solely extreme high/low temperature and precipitation as well as other interactions

² See Isbell et al. (2015) as an example for investigating the insurance effect of species diversity using several datasets.

than high temperature and low precipitation might be important (e.g. Rosenzweig et al. 2002; Schlenker and Roberts 2009; Barlow et al. 2015; Tack et al. 2017) for feed and other agricultural prices and remain an important area for future research. Moreover, more in-depth theoretical and empirical investigations of weather shocks on prices and price dynamics would help to improve farm management and in designing policies. These investigation are even more important given current climate change and the predicted increase of extreme weather events.

1.5 Authors and Author Contributions

Chapter 2 – Economic Benefits from Plant Species Diversity in Intensively Managed Grasslands

- Authors: Sergei Schaub, Nina Buchmann, Andreas Lüscher and Robert Finger
- Authors Contributions: S.S., R.F. and N.B. designed the study. S.S. carried out the statistical analysis and prepared all figures. All authors wrote and commented on this manuscript.

Chapter 3 – Plant Diversity Effects on Forage Quality, Yield and Revenues of Semi-natural Grasslands

- Authors: Sergei Schaub, Robert Finger, Florian Leiber, Stefan Probst, Michael Kreuzer, Alexandra Weigelt, Nina Buchmann and Michael Scherer-Lorenzen
- Authors Contributions: A.W. and M.S.L. designed the original Management Experiment; A.W., M.S.L., F.L., S.P. and M.K. carried out the biomass and forage quality analyses; S.S., R.F. and N.B. designed, and S.S. and R.F. carried out the economic analyses of the Management Experiment; S.S. re-analysed all data; all authors wrote and commented on this manuscript.

Chapter 4 – Economics of Species Diversity in Grasslands under Increasing Drought Risks

- Authors: Sergei Schaub, Nina Buchmann, Andreas Lüscher and Robert Finger
- Authors Contributions: S.S. and R.F. wrote the main manuscript, designed the study and carried out the theoretical analysis. S.S. carried out the statistical analysis and prepared all figures. A.L. provided some of the data used in the manuscript. N.B. and A.L. commented on various versions of the manuscript. All authors reviewed the final manuscript.

Chapter 5 – Drought Effects on Hay and Feed Grain Prices

- Authors: Sergei Schaub and Robert Finger
- Authors Contributions: S. S. and R. F. designed the study. S. S. carried out the data preparation and analysis. S. S. and R. F. wrote the manuscript.

Thesis Appendix 1 – [A Way Of] Exploring our Tolerance Towards Biodiversity Loss and Re-thinking our Biodiversity Communication

- Authors: Sergei Schaub

Thesis Appendix 2 – Global Climate Change and Biodiversity Interest Trends, Research-Policy Efforts and Activism

- Authors: Authors: Sergei Schaub and Robert Finger

- Authors Contributions: S. S. designed the study, carried out the data preparation, analysis and visualization. R.F. provided feedback on the design of the analysis and commented on the conceptual design of the study. S. S. and R. F. wrote the manuscript.

Thesis Appendix 3 – Tracking Societal Concerns on Pesticide – A Google Trends Analysis

- Authors: Sergei Schaub, Robert Huber and Robert Finger
- Authors Contributions: All authors designed the study and wrote the manuscript. S.S. carried out the data preparation, analysis and visualization.

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2 Chapter 2 – Economic Benefits from Plant Species Diversity in Intensively Managed Grasslands³

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Abstract

Grasslands cover a major share of the world's agricultural area and are important for global food security. Plant species diversity in grasslands is known to increase and stabilize biomass yields. We economically evaluate these effects, using a rich dataset from 16 intensively managed grassland sites across Europe. We extend earlier research by accounting for plant species diversity effects on both quantity and quality of yields. Consequently, we can express plant species diversity effects in terms of milk production potential yields per hectare and potential revenues thereof. Plant species diversity not only increased milk production potential yields and thus revenues, but also reduced production risks. Thus, increasing plant species diversity resulted in higher certainty equivalents, for example, the certainty equivalent rose by +29% when comparing the average mixture to the average monoculture. For risk averse decision makers, this gain in certainty equivalent was mainly due to the increase in revenues (accounting for 90%) compared to the total insurance value (accounting for 10%). Overall, we show that farmers benefit economically from plant species diversity and that even a moderate increase in this diversity contributes to more stable grassland-based production. Thus, our results are highly relevant for future sustainable intensification of grassland-based production.

Highlights

- We used a rich dataset from 16 intensively managed grassland sites across Europe
- Plant species diversity increased milk production potential yields, and thus revenue
- Production risks decreased in grasslands with higher plant species diversity
- Plant species diversity constituted a significant insurance value for farmers

Keywords: species diversity, insurance value, risk, stability, sustainable intensification, biodiversity

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2.1 Introduction

Grasslands play a central role in global food security. They cover a major share of the world's agricultural area and are the basis for both forage production and a wide range of additional ecosystem services (Sala and Paruelo 1997). Growing population, changes in consumer demand and climatic challenges increase pressure on grassland-based production. Grassland biomass yields and their quality are affected by site-specific characteristics and farmers' management decisions, for example, with respect to land use intensity. Furthermore, plant species diversity also plays an important role in grasslands. This paper provides an economic evaluation of the effects of plant species diversity (henceforth 'diversity effect') on both quantity and quality of yields as well as production risks using an empirical analysis comprising 16 intensively managed grassland sites across Europe.

Plant species diversity can affect grasslands in three ways: Firstly, plant species diversity increases biomass yields (see e.g. Tilman et al. 1996, Marquard et al. 2009, Finn et al. 2013). This effect of plant species diversity is driven mainly by the complementarity and sampling effects (see e.g. Loreau and Hector 2001, Cardinale et al. 2007). The complementarity effect arises either from greater efficiency in acquisition of available resources, as different species have different needs or/and sources of resources, or from positive interactions between different species. One example for this positive interaction is the nitrogen fixing ability of legumes from which other species in the community also benefit (see e.g. Carlsson and Huss-Danell 2003, Lüscher et al. 2014), particularly in grass-legume mixtures. The sampling effect is based on the increased probability that a community includes highly performing species, which then also become dominant. Furthermore, plant species diverse grasslands suffer less from weed invasion (Suter et al. 2017, Connolly et al. 2018), which reduces both the risk of biomass yield loss of the sown species and the costs of weed control.

Secondly, increasing plant species diversity has been found to increase the stability of biomass yields over time (see e.g. Isbell et al. 2009, Hallett et al. 2017, Haughey et al. 2018). Communities with more species are in a better position to guarantee that some species maintain functioning even when others fail. This is called the 'insurance effect' of plant species diversity in the ecological literature (Yachi and Loreau 1999).

Thirdly, plant species diversity can also influence the quality of biomass yields. However, the reported effects on forage quality and quality corrected yields (biomass yields \times forage quality) are ambiguous (see e.g. White et al. 2004, Deak et al. 2007, Khalsa et al. 2012, Sturludóttir et al. 2014). Furthermore, studies show that large effect sizes of plant species diversity for amount of biomass yields and quality corrected yields (i.e. nitrogen/protein yields) can be achieved already with an increase of species numbers from one to four species provided that legumes are combined with grasses (Kirwan et al. 2007, Finn et al. 2013, Suter et al. 2015). Implementing such moderate increases in plant species diversity can be considered already now as a 'ready to use' strategy for sustainable intensification.

Existing agricultural and ecological economic research shows that farmers benefit from both higher and more stable biomass yields (Baumgärtner 2007, Baumgärtner and Quaas 2010). However, only a few studies have monetarized the diversity effect on grassland yields and its stability (Schläpfer et al. 2002, Koellner and Schmitz 2006, Finger and Buchmann 2015, Binder et al. 2018). These studies show that grasslands with a higher plant species diversity generate increases in farmers' expected utility. However, current economic literature has four major shortcomings: Firstly, past studies have been restricted to single or a small number of sites. Secondly, earlier studies were usually limited to (very) extensively managed grasslands. Therefore, the implications of these studies for real dairy farm-level decision making remain limited. Thirdly, existing economic literature has not economically evaluated the potential impact of the diversity effect on quality corrected yields. However, accounting for forage quality is crucial for the performance of the production system as it determines the potential to produce meat and milk (e.g. Briner et al. 2015). As an exception, Binder et al. (2018) economically assessed the diversity effect on crude protein contents, in addition to the plant species diversity-biomass yields relationship. However, crude protein alone is insufficient to monetarize the impact of plant species diversity on quality corrected yields because (metabolizable) energy is usually the first restricting factor for ruminant production (Barnes et al. 2003). Fourthly, the effects of plant species diversity on the variability of quality corrected yields have not been addressed so far.

This study contributes to close these gaps by evaluating the diversity effect from an agricultural economic perspective. In this paper, we (1) investigate the mean response of biomass yields, i.e. dry matter of biomass yields (DM yields; kg ha^{-1}), forage quality, i.e. milk production potential per kg of DM yield (DM MPP; $\text{kg kg}_{\text{DM}}^{-1}$) and quality corrected yields, i.e. milk production potential yields (MPP yields; kg ha^{-1}), to altered plant species diversity levels in intensively managed grassland sites across a wide range of pedo-climatic conditions; (2) quantify effects of plant species diversity on the variability of biomass yields, forage quality and quality corrected yields. Thus, we quantify effects on production risks; (3) economically evaluate species diversity effects using certainty equivalents and stochastic dominance; (4) test whether the diversity effect is persistent when the best performing monocultures are compared with all mixtures and with the best performing mixtures. For our analysis, we use a dataset that comprises information from 16 experimental sites across Europe on grass monocultures, legume monocultures and grass-legume mixtures with four functionally distinct species (Kirwan et al. 2014).

The remainder of this paper is organized as follows. In Section 2, we develop the ecological-economic and econometric framework. The data and measurement of the data are described in Section 3, followed by results in Section 4. Finally, we discuss and present our conclusions in Section 5.

2.2 Methodological Framework

2.2.1 Agricultural Economic Valuation of Uncertain Outcome

Farmers' production decisions influence forage production. In grassland-based production systems, forage production can be influenced by plant species diversity, D , as well as by other management and environmental factors, X , which have consequences on biomass yields, forage quality and quality corrected yields. In this paper, we use the variable dry matter yields (DM yields) for biomass yields, and milk production potential per kg of dry matter yield (DM MPP) for forage quality as only higher quality can lead to higher milk production per unit dry matter. Finally, we use the variable milk production potential yields (MPP yields) for quality corrected yields (MPP yields $(D,X) = \text{DM yields } (D,X) \times \text{DM MPP } (D,X)$). The use of MPP yields allows a direct link between production and farm revenues, π :

$$\pi (D,X) = \text{MPP yields } (D,X) \times p \quad (1)$$

where p is the price of milk. The inherent variability of MPP yields causes revenues to be stochastic. This variability is a function of plant species diversity as well as other management and environmental factors.

We use an expected utility framework where plant species diversity is the decision variable. In this framework, the implicit costs arising from risk exposure (i.e. variability of revenues) are captured in the risk premium, RP (see e.g. Chavas 2004). The difference between the expected stochastic revenues $E(\pi(D, X))$ and the risk premium $RP(D, X)$ is equal to the utility arising from a deterministic payment, the so called certainty equivalent (CE):

$$CE = E(\pi(D, X)) - RP(D, X) \quad (2)$$

where $E(\cdot)$ is the expectation operator. The risk premium can be approximated as follows (Pratt 1964):

$$RP \approx 0.5 r \text{Var}(\pi(D, X)) \quad (3)$$

where r refers to the Arrow-Pratt risk coefficient of absolute risk aversion, which indicates the level of risk aversion of an individual. The coefficient is defined as: $r = -U''/U'$, where U' and U'' represent first and second derivatives of the utility function $U(\cdot)$, respectively. The relevance of the latter depends on the subjective risk preferences of individual decision makers. We assume farmers are risk averse (see e.g. Maart-Noelck and Musshoff 2014, Meraner and Finger 2018, Iyer et al. 2019). In the following, we use a relative coefficient of risk aversion of 2, if not mentioned otherwise, which represents rather risk averse behavior (Hardaker et al. 2015).

$\text{Var}(\pi(D, X))$ is the variance of the revenues and reflects that we expect plant species diversity (D) and other management and environmental factors (X) to affect the variability of quality corrected yields. We focus on deterministic price levels, so that the variance of the revenues can be expressed as $\text{Var}(y(D, X)) p^2$. Thus, the risk premium is defined as follows:

$$RP = 0.5 r \text{Var}(\pi(D, X)) = 0.5 r \text{Var}(y(D, X)) p^2 \quad (4)$$

In turn, the calculation of CE is as follows:

$$CE = E(\pi(D, X)) - 0.5 r \text{Var}(y(D, X)) p^2 \quad (5)$$

The insurance value of plant species diversity, $IV(D)$, is the negative of the marginal effect of plant species diversity on the risk premium (Baumgärtner 2007, Finger and Buchmann 2015):

$$IV(D) = -\partial RP / \partial D = -0.5 r p^2 \partial \text{Var}(y(D, X)) / \partial D \quad (6)$$

Thus, $IV(D)$ describes how plant species diversity reduces the cost of risks borne by farmers and allows us to monetarize the risk altering property of plant species diversity from a farmer's utility perspective. Finally, we derive the total insurance value (total IV) at a certain plant species diversity level by:

$$\text{total IV} = \int_0^D IV(D) dD = \int_0^D (-(-0.5 r p^2 \partial \text{Var}(y(D, X)) / \partial D)) dD \quad (7)$$

The CE requires precise information (or assumptions) about the coefficient of risk aversion r . This limitation is overcome by complementing the CE-based analysis with stochastic dominance analysis to obtain a more general basis for comparison without requiring exact information about risk aversion (see e.g. Chavas 2004). The stochastic dominance inference is based on a binary comparison, for example of plant species diversity \underline{a} with plant species diversity \underline{b} . Choice a is preferred over choice b if the utility arising from the respective probability density function of revenues is larger:

$$E_a U(\pi) \geq E_b U(\pi) \quad (8)$$

Applying first order stochastic dominance (see e.g. Chavas 2004), choice a dominates choice b when the underlying cumulative distribution functions, $A(\pi)$ and $B(\pi)$ respectively,⁴ follow:

$$A(\pi) \leq B(\pi) \text{ for all } \pi \quad (9)$$

If choice a first order dominates choice b , the cumulative distribution function of choice a is always beneath and right of the cumulative distribution function of choice b . Thus, the first order stochastic dominance criterion is independent of risk preferences and only requires that $U' > 0$. However, the

⁴ The cumulative distribution function is defined as the integral of its probability density function, e.g. for a : $A(\pi) = \int_{-\infty}^{\pi} a(t) dt$.

discriminatory power of the criterion is often low. If cumulative distribution functions cross, further stochastic dominance criteria can be used, such as the second order stochastic dominance (see e.g. Chavas 2004), which is defined as:

$$\int_{-\infty}^{\pi} A(t)dt \leq \int_{-\infty}^{\pi} B(t)dt \text{ for all } \pi \quad (10)$$

Thus, choice a dominates choice b when the total area beneath the cumulative distribution function of choice a at every level of π is smaller than the total area beneath the cumulative distribution function of choice b. The second order stochastic dominance criterion implies that the decision maker is risk averse, i.e. $r > 0$.

A major limitation of stochastic dominance criteria is that outcomes of these assessments for a specific sample are binary, i.e. that one choice (e.g. a specific level of plant species diversity) either dominates another or not. Thus, this approach remains inconclusive in many applications and cannot account for uncertainties underlying the sample composition and results might be driven by specific observations in the sample. In order to overcome these limitations and allow for statistical inference, we apply a test procedure based on a simulated Kolmogorov Smirnov test (Barrett and Donald 2003).⁵ For this procedure, we use the 'simulation method 1' provided by Barrett and Donald (2003). The null-hypothesis that choice a dominates choice b, can be rejected when the $p\text{-value}_{a,b} < \alpha$. α represents significance levels of 0.1, 0.05 and 0.01. To obtain a conclusive inference that choice a dominates choice b, we need simultaneously $p\text{-value}_{a,b} > \alpha$ and $p\text{-value}_{b,a} < \alpha$. Note that we do not consider stochastic dominance criteria of higher orders because their additional discriminatory power is expected to be low (Hardaker et al. 2015). Furthermore, it must be borne in mind that empirical stochastic dominance testing does not allow controlling for other influences than plant species diversity, such as for location of the site or year of the experiment. However, to account for key differences in productivity across sites, we only use observations from sites that cover all levels of our diversity gradient in the stochastic dominance analysis.⁶

2.2.2 Econometric Implementation

We use the stochastic production function framework proposed by Just and Pope (1978) to identify the diversity effect on the expected outcome (including biomass yields, forage quality and quality corrected yields) and the variance of outcome.

⁵ The Kolmogorov-Smirnov simulation allows a comparison of revenues at all revenue values, and can be applied to compare different sample sizes. We can also use it to derive statistical inference, as we can estimate p-values in finite samples. This is also true for second order stochastic dominance, the test statistic of which does not have a closed form solution.

⁶ Note that results of all sites for the stochastic dominance analysis can be found in Table A4 and Fig. A3

Firstly, the expected outcome can be estimated by the stochastic production function, which is specified as:

$$y_{i,k} = \alpha + \beta_1 D_i^{0.5} + \beta_2 \text{Site}_i + \beta_3 \text{Density}_i + \beta_4 \text{Year}_i + \beta_5 \text{Site}_i \times \text{Cuts}_i + e_{1,i,k} \quad (11)$$

where $y_{i,k}$ represents either annual biomass yields, forage quality or quality corrected yields depending on the index k across the different years for each plot, i . In equation (11) $y_{i,k}$ comprises all monocultures and all mixtures. We selected a square root specification of plant species diversity, $D_i^{0.5}$, because this allows for a decreasing diversity effects (see e.g. Hooper et al. 2005, Finn et al. 2013) and when compared to other empirical model specifications,⁷ it had the lowest Akaike information criterion (AIC) for annual biomass yields and quality corrected yields. To account for farmers' choices for sowing monocultures or mixtures of different plant species diversity levels and to avoid endogeneity problems in the estimation, we use sown plant species diversity levels to measure D_i . Moreover, we introduce site dummies, Site_i , that account for structural differences in productivity across sites due to the location of the experimental sites. In addition to these site dummies, we include a set of explanatory variables, consisting of a dummy for high/low sown density, Density_i , dummies for the year of the experiment, Year_i , and an interaction term of site and number of cuts per year, $\text{Site}_i \times \text{Cuts}_i$, for those sites with varying cuts per year. The error term, e_1 , comprises all uncontrolled factors, such as the variability of weather and pest infestations. Note that due to the experimental nature of our data, these components of the error term are uncorrelated with the explanatory variables. We assume that $E(e_{1,i,k}) = 0$. Furthermore, we account for structural differences in the variability of the outcome variable across sites, i.e. the standard error is not only influenced by the individual observation when estimating biomass yields, forage quality and quality corrected yields, but also by a common site effect (common cluster effect). To this end, we follow Wooldridge (2003) and model the error term for equation (11) as follows:

$$e_{1,g,i,k} = v_{1,g,k} + z_{1,g,i,k} \quad (12)$$

The subscript g refers to the cluster, i.e. the site, k to either biomass yields, forage quality or quality corrected yields, and i to the single plot. If this clustered structure is ignored in an ordinary least squares (OLS) regression it can lead to a strong bias of standard errors (Moulton 1986). We compute cluster-robust standard errors to correct these biases. This correction also accounts for the expected heteroscedasticity, i.e. the fact that the variance of residuals is bigger with lower levels of plant species diversity.

⁷ Other empirical model specifications include D_i , $D_i + D_i^2$ and $D_i + D_i^{0.5}$.

Secondly, the variance of outcome $y_{i,k}$ is defined as $\text{Var}(y_{i,k}) = (y_{i,k} - \bar{y}_{i,k})^2 = e_{1,i,k}^2$ (Just and Pope 1978) and econometrically specified as:

$$\text{Var}(y_{i,k}) = \alpha + \beta_1 D_i^{0.5} + \beta_2 \text{Site}_i + \beta_3 \text{Density}_i + \beta_4 \text{Year}_i + \beta_5 \text{Site}_i \times \text{Cuts}_i + e_{2,i,k} \quad (13)$$

For this equation, we use again a square root specification of plant species diversity (see e.g. Hooper et al. 2005). The square root specification was chosen since it performed best among the empirical model specifications based on the AIC. We again compute cluster-robust standard errors for this estimation. Note we estimate all models separately using Stata 15.0 for Windows. The Stata code for the econometric estimations as well as the R (R Core Team 2018) code for data preparation are available in the online Appendix.

2.2.3 Comparison of Best Performer Type I & II

A potential limitation when analyzing experimental data is that comparisons of all mixtures with all monocultures lack practical implications for farmers as they want to select only the best monoculture and mixture. This limitation is addressed in two ways. Firstly, the design of the underlying experiment focusses exclusively on high performing species. This reduces the importance of the sampling effect, which only plays a minor role in intensive agricultural systems. Secondly, in addition to the analysis in Section 2.2, we perform comparisons of all mixtures to the best performing monoculture as well as comparisons of the best mixture to the best monoculture. In the following, the former is referred to as comparison of best performer of *type I* and the latter of *type II*. Both comparisons of best performer assume that farmers know the best performing monoculture and/or mixture in advance.

Prior to these best performer analyses, we conduct a pre-test as proposed by Schmid et al. (2008). More specifically, we test if observations within a group are significantly different, i.e. if monocultures differ from each other and if mixtures differ from each other in terms of quality corrected yields. Only if within group differences exists, it makes sense to select the best performer of this group for our analysis. Moreover, to avoid a sampling bias in this analysis, we follow Schmid et al. (2008) and use the mean of all replicates per site and year, with replicates referring to plots with different sown densities and the same sown composition of species (see data description for details). Sown composition includes information about the presence and evenness of each species on a plot. The pre-test commences with a regression analysis, once for all monocultures and once for all mixtures, with quality corrected yields as the dependent variable and with sown compositions, site, year of experiment and cuts per year for sites with varying cuts per year as explanatory variables. Secondly, we conduct Wald tests to analyze whether the coefficients of the sown composition of a regression differ from each other.

Three different rules are applied to identify the best performer per site. Firstly, we assume that farmers will choose those options (monocultures/mixtures) that maximize the average quality corrected yields. Secondly, farmers are assumed to select the monoculture/mixture with the highest minimum quality corrected yields in one year (maximin rule). Thirdly, farmers select the monoculture/mixture with the maximal possible quality corrected yields in one year (maximax rule). The identification of the best performer is based on the average quality corrected yields of the replicates.⁸

The estimation of the diversity effect of the type I comparison differs from equation (11) as we do not control for sown density because the best performer is identified by the mean of the replicates, i.e. plots with same sown composition but different sown densities:

$$y_{i,k} = \alpha + \beta_1 D_i^{0.5} + \beta_2 \text{Site}_i + \beta_3 \text{Year}_i + \beta_4 \text{Site}_i \times \text{Cuts}_i + e_{3,i,k} \quad (14)$$

Note the index k comprises only annual quality corrected yields across different years in this comparison of best performer. The dataset for this analysis includes all mixtures but only the best performing monocultures.

For type II, the model is adjusted as follows:

$$y_{i,k} = \alpha + \beta_1 \text{Mixture}_i + \beta_2 \text{Site}_i + \beta_3 \text{Year}_i + \beta_4 \text{Site}_i \times \text{Cuts}_i + e_{4,i,k} \quad (15)$$

In this model, k comprises only annual quality corrected yields across different years. Furthermore, we use a mixture dummy, Mixture_i , as we only have one mixture per site (the best performing mixture), thus, there is no meaningful gradient of the diversity gradient. The data for the type II comparison consists exclusively of the best performing mixtures and the best performing monocultures. Both estimations, equations (14) and (15), are corrected for clustered error terms and heteroscedasticity as described above.

2.3 Data – Forage Quantity & Forage Quality

2.3.1 Experimental Setup

The analysis is based on biomass yields and forage quality data retrieved from the database of the *COST Agrodiversity Experiment* (Kirwan et al. 2014), which is available online. A subset of this dataset was used here, including data from 16 intensively managed grassland sites (8 European countries,

⁸ General selection rules determining how farmers choose the best grassland monoculture/mixture comprise opting for the highest profits, the highest yields, the highest utility, or simply copying the status quo (Huber et al. 2018). In practice, decisions about which plants to sow must be taken on the basis of past experience and before the actual yield is known, whereas our selection rules are based on actual yields. Hence, our best performer comparison of type I is more conservative, as it selects the monoculture that really performed the best. Furthermore, seed costs of monocultures and mixtures in the study are very low compared to other costs and management costs for monocultures and mixtures are fairly similar. Thus, these costs do not drive farmers' decisions.

Figure 1). The coordinated experiment was carried out between 2003 and 2011 and lasted at each site between 2 to 4 years. However, forage quality data was not collected in all years at some sites (see Table A1 for details). The specific management regimes at each site represented intensive management adopted to the local condition. Over all sites, the management intensity ranged between 2 and 5 cuts and between no fertilizer and 150 kg ha⁻¹ nitrogen, 70 kg ha⁻¹ phosphorus and 420 kg ha⁻¹ potash fertilizer annually. In general, management intensity did not vary within a site, except for three sites at which the numbers of cuts per year varied between years (Table A1). The experiment comprised four monocultures and eleven mixtures of four functional types of species that produce forage of high quality in intensively managed systems. The eleven mixtures with low species numbers contained four species in different sown proportions (four dominated mixtures with 0.7, 0.1, 0.1, 0.1 in turn, six co-dominated mixtures with 0.4, 0.4, 0.1, 0.1 in turn, and one equiproportional mixture with 0.25, 0.25, 0.25, 0.25; Kirwan et al. 2014). The monoculture and mixtures thus reflect a plant species diversity gradient (*sensu* Isbell et al. 2015, Connolly et al. 2018, Finn et al. 2018). These diversity levels depend on the plant species richness (number of species) and their evenness (relative abundance) in the sward and can be expressed by the Simpson index of diversity (Krebs 1999). The Simpson index is frequently used and considers species richness and evenness:

$$\text{Simpson index} = 1 - \sum(h_i)^2 \quad (15)$$

where h_i is the proportion of individuals of species i in the community. The respective values of the Simpson index of sown plant species diversity in the experiment are: 0.48, 0.66 and 0.75 for the different mixtures and 0 for monoculture. The functional types of the four species were selected in order to achieve large functional differences among them, and consisted of a fast-establishing grass, a slow-establishing persistent grass, a fast-establishing legume and a slow-establishing persistent legume. The same four were used in each mixture (Kirwan et al. 2014). Each sward was sown in two density levels. For more information about the experimental design, see Kirwan et al. (2014).



Figure 1: Location of experimental sites used in this study. Numbers indicate the site indications according to Kirwan et al. (2014). See Table A1 for details on the sites and mixture types.

2.3.2 Data Collection & Measurement

DM yields (kg ha^{-1}) were determined for each plot and harvest (Kirwan et al. 2014). For evaluating and monetarizing forage quality, we focused on DM MPP ($\text{kg kg}_{\text{DM}}^{-1}$). DM MPP describes the potential milk produced per DM yield and it was derived from metabolizable energy (DM ME; $\text{MJ kg}_{\text{DM}}^{-1}$) and ash content ($\text{g kg}_{\text{DM}}^{-1}$; Gierus et al. 2012, Jans et al. 2015):

$$DM\ MPP = DM\ ME \times [0.46 + 12.38 \times DM\ ME / (1000 - ash)] / 3.14 \quad (17)$$

DM ME is the energy content per DM yield available for maintenance, milk production and weight gains and is used to assess overall ruminant-specific nutritive value. (Metabolizable) Energy is usually the factor of forage that is first limiting in ruminant production (Barnes et al. 2003). DM ME and ash content ($\text{g kg}_{\text{DM}}^{-1}$; ash consists mainly of minerals) were obtained by using near infrared reflectance spectroscopy (NIRS). DM yields were multiplied with DM MPP to compute MPP yields (kg ha^{-1}):

$$MPP\ yields = DM\ yields \times DM\ MPP \quad (18)$$

Moreover, the annual DM yields and MPP yields were calculated by tallying up all harvests per year.⁹ The average DM MPP of a year was calculated by taking the mean DM MPP of all harvests weighted by DM yields of the harvests. See Kirwan et al. (2014) for more details about data collection. Finally, the annual revenues from milk sales were calculated by multiplying annual MPP yields with the milk

⁹ Note for site 10 quality was only measured for the first four out of five cuts each year. As this is consistent for all plots and the first cuts are agronomically more important, these observations are included in the analysis.

price of 0.35 Euro $\text{kg}_{\text{milk}}^{-1}$ (EU average of the years 2013 to 2015; Eurostat 2017).¹⁰ Furthermore, results of DM ME and ME yields are available in the online Appendix.

The summary statistics in Table 1 shows that DM yields and MPP yields have a mean of 8748 kg ha^{-1} and 18750 kg ha^{-1} across all observations. The mean for both is lower for monocultures and higher for mixtures. In contrast, mean DM MPP is comparable between monocultures and mixtures. DM yields and MPP yields vary considerably among sites, while DM MPP is more similar across sites (Fig. A1 to A3).

Table 1: Summary statistics: mean (and standard deviation) of annual dry matter yields (DM yields), milk production potentials per kg of DM yield (DM MPP) and milk production potential yields (MPP yields).

	All observations	Monocultures	Mixtures
DM yields (kg ha^{-1})	8748 (3886)	7235 (3519)	9614 (3824)
DM MPP ($\text{kg kg}_{\text{DM}}^{-1}$)	2.15 (0.09)	2.15 (0.10)	2.14 (0.09)
MPP yields (kg ha^{-1})	18750 (8599)	15500 (7759)	20610 (8512)

2.4 Results

In this section, we present first the empirical results of the diversity effect on expected outcomes, including dry matter yields (DM yields), milk production potentials per kg of DM yield (DM MPP), and milk production potential yields (MPP yields), and their respective variance (4.1). Next, we use these empirical results for computing the certainty equivalents, expected revenues as well as total insurance values and we perform the stochastic dominance analysis (4.2). In the end of this section, we conduct the comparison of best performer (4.3).

2.4.1 Diversity effects on DM Yields, DM MPP & MPP Yields

Our analysis showed a positive but diminishing effect of plant species diversity on DM yields and MPP yields, while the effect on DM MPP was not significant (Table 2).¹¹ DM yields and MPP yields increased by about 25% when comparing the average mixture (Simpson index = 0.6)¹² with the average monoculture. Furthermore, plant species diversity reduced production risks, as the variance of all three outcomes decreased (Table 2). More specifically, the diversity effect on the variance of DM yields

¹⁰ In addition, we conducted a sensitivity analysis covering different milk prices, ranging from the lowest to the highest EU average milk prices (2013-2015) of a country with a site in our data: 0.27 to 0.36 Euro $\text{kg}_{\text{milk}}^{-1}$.

¹¹ Note that results of DM ME and ME yield are similar to the results of DM MPP and MPP yield and can be found in Table A3.

¹² Diversity effects for the average mixture represents the diversity effects at the mean Simpson index value of all mixtures, which is 0.6.

was significantly negative, while the diversity effect on the variance of DM MPP was negative but not significant. The overall diversity effect on the variance of MPP yields was also negative and significant at the 10% level.

Table 2: Results of the effect of plant species diversity on dry matter yields (DM yields), milk production potentials per kg of DM yield (DM MPP) and milk production potential yields (MPP yields; equation 11) and their variance (equation 13).

Expected outcome		Variance of outcome	
DM yields (kg ha ⁻¹)		Var(DM yields)	
Simpson index ^{0.5} (D ^{0.5})	2599 (6.24)***	Simpson index ^{0.5} (D ^{0.5})	-2.79×10 ⁶ (-2.41)**
R ² _{adj}	0.766	R ² _{adj}	0.157
DM MPP (kg kg _{DM} ⁻¹)		Var(DM MPP)	
Simpson index ^{0.5} (D ^{0.5})	-0.02 (-1.31)	Simpson index ^{0.5} (D ^{0.5})	-0.011 (-1.16)
R ² _{adj}	0.465	R ² _{adj}	0.025
MPP yields (kg ha ⁻¹)		Var(MPP yields)	
Simpson index ^{0.5} (D ^{0.5})	5423.78 (6.32)***	Simpson index ^{0.5} (D ^{0.5})	-1.13×10 ⁷ (-1.95)*
R ² _{adj}	0.787	R ² _{adj}	0.161

* p < 0.1; ** p < 0.05; *** p < 0.01. Numbers in parentheses are t-values. T-values are corrected for clustered error terms and heteroscedasticity. Number of observations = 698. Adjusted R² refer to the full models and all the coefficients of the models are available in the online Appendix.

2.4.2 Certainty Equivalent & Stochastic Dominance

The expected revenues, total insurance value, and thus CE increased with plant species diversity (Fig. 2). The rates of increase were lower at higher levels of plant species diversity. Due to the positive effect of plant species diversity on CE, we report a positive effect of plant species diversity on farmers' utility. Overall, the diversity effect increased the CE for the average mixtures (Simpson index = 0.6) by about 1630 Euro ha⁻¹ compared to the average monocultures for rather risk averse farmers (r = 2). In relative terms, this is a gain of about 29%. To a large extent, the increase was due to a gain in expected revenues of about 1470 Euro ha⁻¹ and to a lesser extent to a gain in total insurance value of about 160 Euro ha⁻¹. Re-running the analysis with varying coefficients of risk aversion, from low risk aversion (r = 0.5) to very high aversion (r = 4; Hardaker et al. 2015), resulted in a total insurance value for the average mixtures between about 40 to 330 Euro ha⁻¹ (Fig. 3). The relative values for this range of total insurance values compared to expected revenues of the mean mixture are about 3% and 22%.

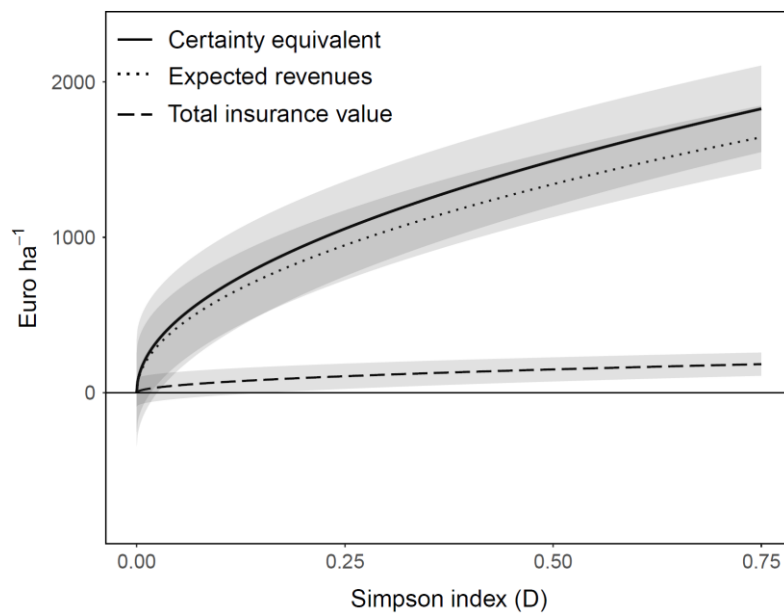
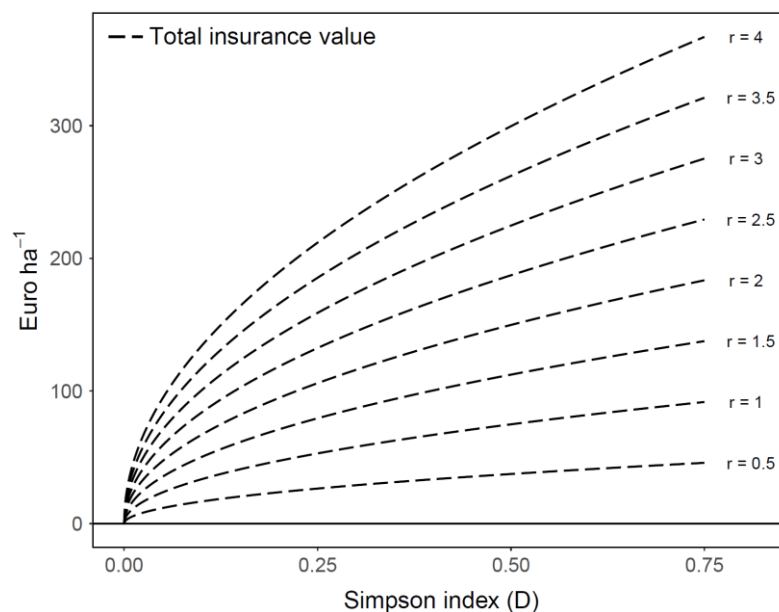


Figure 2: Effect of plant species diversity on the certainty equivalent (CE), expected revenues and total insurance value in Euro ha⁻¹ for rather risk averse farmers.¹³

Note: The values show the species diversity effect for the mean site and a coefficient of risk aversion of 2. The diversity effect represents the additional gain from species diversity compared to the average monoculture. CE = expected revenues + total insurance value. The 90-percent confidence intervals are based on standard errors that are corrected for clustered error terms and heteroscedasticity.



¹³ Assuming milk prices ranging from 0.27 to 0.36 Euro kg_{milk}⁻¹, the CE, the expected revenues and the total insurance value for the average mixtures compared to a monoculture range from ~1260 to ~1680 Euro ha⁻¹, from ~1130 to ~1510 Euro ha⁻¹ and from ~130 to ~170 Euro ha⁻¹, respectively. More detailed results are available upon request from the authors.

Figure 3: Effect of plant species diversity on total insurance value in Euro ha⁻¹ for a range of risk aversions of farmers. Increasing coefficients r represent increasing risk aversion of farmers.

Concerning stochastic dominance, we found that the tested mixtures (Simpson index 0.48, 0.66 and 0.75) dominate the monocultures (Simpson index = 0) in a first and second order stochastic dominance sense. This is illustrated in Figure 4, where the cumulative distribution function of monocultures is always to the left of the cumulative distribution functions of mixtures. This dominance of mixtures was confirmed by the empirical Kolmogorov Smirnov test as for all mixtures the $p\text{-value}_{\text{Mixtures, Monocultures}} > 0.1$ and the $p\text{-value}_{\text{Monocultures, Mixtures}} < 0.1$. Regarding dominance across mixtures, we found no first or second order dominance.

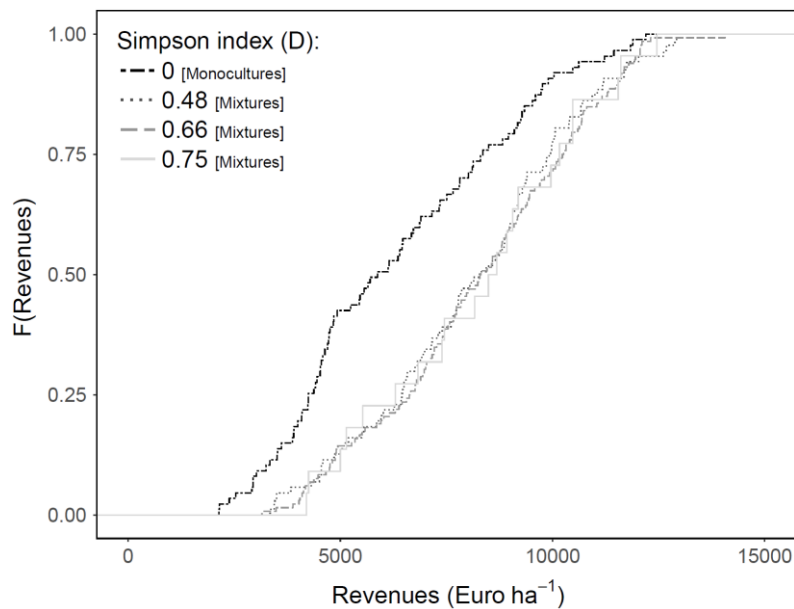


Figure 4: Cumulative distribution function of milk revenues per Simpson index level. $F(\text{Revenue})$ indicates the probability of the observed revenue falling below a certain level.

Note: Observations are only from sites that comprise the entire range of the Simpson index. Number of sites = 6. Number of observations = 328.

Table 3: Results of the empirical Kolmogorov Smirnov test for stochastic dominance of first and second order. Binary comparison of swards with different Simpson indices (0 = monocultures; 0.48, 0.66 and 0.75 = mixtures).

Simpson index (D)		First order stochastic dominance		Second order stochastic dominance	
A	B	p-value _{a,b}	p-value _{b,a}	p-value _{a,b}	p-value _{b,a}
0.48	0	1	<0.001	0.8	<0.001
0.66	0	1	<0.001	0.4	<0.001
0.75	0	1	0.018	0.8	<0.001
0.66	0.48	0.857	0.293	0.4	0.2
0.75	0.48	0.936	0.481	1	0.2
0.75	0.66	0.732	0.77	0.6	0.4

Note: The null-hypothesis, that for example choice a dominates choice b, cannot be rejected if the $p\text{-value}_{a,b} > \alpha$. Observations are only from sites that comprise the entire range of the Simpson index. Number of sites = 6. Number of observations = 328.

2.4.3 Comparison of Best Performer Type I & II

Additional to the above analysis, i.e. all mixtures vs. all monocultures, we conducted a comparison of best performer of type I, i.e. all mixtures vs. the best monocultures, and type II, i.e. the best mixtures vs. the best monocultures. We did this because the pre-test revealed significant differences within groups (Wald tests for differences within monocultures: $p\text{-value}=0.02$ and for differences within mixtures: $p\text{-value}<0.001$).

The comparison of best performer of type I showed that, regardless of the rule defining how the best monocultures were selected, plant species diversity had a positive effect on MPP yields (Table 4). Moreover, MPP yields of mixtures were also significantly higher in the comparison of best performer of type II for all rules (Table 4). Here, a mixture dummy instead of the square root of the Simpson index captures the diversity effect. The diversity effect was lowest when employing the maximize average MPP yields rule and the maximax rule, and was highest when using the maximin rule. Overall, the effect of plant species diversity was smaller in the best performer comparisons of type I and II than it was when all monocultures and all mixtures were compared (see Table 2).

Table 4: Results of the comparison of best performer of type I and type II with milk production potential yields (MPP yields; kg ha⁻¹) as the dependent variable.

Rule	Type I		Type II	
Maximize average MPP yields	Simpson index ^{0.5} (D ^{0.5})	1068.36 (2.06)*	Mixture dummy	2324.81 (4.84)***
	Adjusted R ²	0.849	Adjusted R ²	0.759
Maximin rule	Simpson index ^{0.5} (D ^{0.5})	1891.79 (2.37)**	Mixture dummy	2648.18 (3.20)***
	Adjusted R ²	0.843	Adjusted R ²	0.755
Maximax rule	Simpson index ^{0.5} (D ^{0.5})	1068.36 (2.06)*	Mixture dummy	2291.50 (4.57)***
	Adjusted R ²	0.849	Adjusted R ²	0.753

* p < 0.1; ** p < 0.05; *** p < 0.01. Numbers in parentheses are t-values. T-values are corrected for clustered error terms and heteroscedasticity. The rules maximize average MPP yields and maximax lead to the selection of the same best monocultures. Number of observations: type I = 473, type II = 64.

2.5 Discussion & Conclusion

We found that plant species diversity as a production factor in intensively managed grasslands was beneficial for milk production per area of land as already a moderate increase in plant species diversity (up to 4 species) increased and stabilized quality corrected yields, i.e. MPP yields. Khalsa et al. (2012) also showed a positive impact of plant species diversity on energy related yields, i.e. gross energy, in extensively managed grasslands. In the case of biomass yields, i.e. DM yields, our results that grasslands with higher plant species diversity produce more DM yields support findings from earlier studies on both intensively managed grasslands (e.g. Finn et al. 2013¹⁴) and extensively managed grasslands (up to 60 plant species; e.g. Marquard et al. 2009; Tilman et al. 1996). Plant species diversity did not affect forage quality, i.e. DM MPP, which is similar to results of earlier studies that found insignificant to small effects (Deak et al. 2007, Khalsa et al. 2012, Sturludóttir et al. 2014). However, our results on forage quality differed from White et al. (2004), who found a negative relationship between plant species diversity and forage quality. The latter study, however, did not control for differing environmental conditions of grasslands with low and high plant species diversity. We can also confirm former findings concerning the positive stability impact of plant species diversity on DM yields

¹⁴ Note that we employed a subset (16 out of 31 sites) of the biomass data used by Finn et al. (2013).

for intensively managed grasslands (e.g. Isbell et al. 2009, Hallett et al. 2017, Haughey et al. 2018, Wang et al. 2019). Moreover, we found no significant diversity effects on the variability of DM MPP. As a result, MPP yields were more stable at higher plant species diversity, which consequently implies lower production risks for farmers. Therefore, we can show a significant insurance effect of plant species diversity not only for DM yields but also for MPP yields, even when considering a moderate increase in plant species diversity.

We used two complementary approaches to assess the economic value of plant species diversity from the perspective of a risk averse farmer, i.e. certainty equivalents and stochastic dominance. Increased plant species diversity implies considerable monetary benefits for farmers in terms of CE. The CE benefits amounted to a large part from an increase in the expected revenues and to a lesser degree to an increase in the total insurance value of plant species diversity. In the case of rather risk averse farmers ($r = 2$), the respective values were about 1470 and 160 Euro ha⁻¹ for the average mixtures compared to the average monoculture. Thus, our results show that farmers gain from plant species diversity already at low levels of species diversity. Therefore, farmers have an incentive to increase plant species diversity until the resulting costs exceed the expected increase in CE. The optimal level of plant species diversity increases with farmers' risk aversion. Costs of plant species diversity were not considered in our analysis as the costs of experiments cannot serve as a proxy for farm settings. Stochastic dominance also supported the CE inference that mixtures were preferred over monocultures. However, we could not determine that mixtures with higher levels of plant species diversity dominated mixtures with lower levels. This might be due to the fact that stochastic dominance analysis does not allow controlling for any independent variables other than plant species diversity or to the diminishment of the diversity effect and that our mixtures only differ in evenness. The latter would indicate that in agricultural practice, the presence of different species is more important than the exact distribution of each.

Finally, the comparison of best performer of type I and II also showed that grasslands with greater plant species diversity generated higher MPP yields. These additional comparisons were more conservative tests than comparing all mixtures with all monocultures as these comparisons assume that farmers know swards' performances. However, farmers might only be able to select good, but not the best species, as Finn et al. (2013) showed that the best species often changes across years (in 26 out of 54 cases). Thus, our comparison of all mixtures with all monocultures from a pool of four high performing species, reflects decision making in agricultural practice. In addition, the uncertainty about which species will produce the best performance in a certain year is a strong argument in favor of mixtures. This is because the increased number of species in the sward ultimately increases the probability of it containing the best performing species, leading to higher and more stable quality corrected yields.

Our findings on risk reducing properties and possible economic benefits of plant species diversity in grasslands are in line with earlier research (Schläpfer et al. 2002, Koellner and Schmitz 2006, Finger and Buchmann 2015, Binder et al. 2018). Our results go beyond these earlier findings and show that this also holds for quality corrected yields, i.e. MPP yields, a critical variable for farmers' decision making, in intensively managed grasslands. Moreover, our results are more robust and transferable into real world settings because we use data from a wide range of pedo-climatic conditions across Europe. In addition, the plant species in the experiment are suitable for intensive management and they meet quality demands of lactating cows. Thus, our findings are highly important for farmers, extension services and policy makers. Policy makers and extension services for example can use our results about the plant species diversity benefits, expressed in economic terms, to encourage farmers to increase plant species diversity in grasslands. Furthermore, the mixtures used in our analysis are mixtures with low species numbers readily available for application in agricultural practice. There are several ways to increase plant species diversity, such as over-sowing existing swards with additional species, drilling of seed mixtures into weed free plots, or sowing ex-arable fields with mixtures (see e.g. Walker et al. 2004). Furthermore, the risk reduction property of plant species diversity, even at low plant species diversity and when considering quality corrected yields, supports that plant species diversity can substitute financial insurances (Baumgärtner 2007). This property is crucial, especially because insurance mechanisms are often not available for grasslands (see e.g. Vroege et al. 2019). Furthermore, the positive insurance value is likely to increase in importance under more variable and uncertain future environmental conditions. Thus, our findings contribute more general to better risk management in dairy production (e.g. Finger et al. 2018). There are other positive effects of plant species diversity which are not included in our valuation, such as supporting pollinators, providing weed control and nitrogen fixation (Potts et al. 2009, Suter et al. 2015, Suter et al. 2017, Connolly et al. 2018). Our results show that private and public goal functions are not in conflict to each other, but increased plant species diversity may induce private and public welfare gains. Policy makers could further promote these positive diversity effects by providing incentives for more plant species diverse temporal grasslands. Baumgärtner and Quaas (2010) show for example that subsidies can be efficient policy measures to increase plant species diversity and provide private and public benefits.

In conclusion, our results show that already a moderate increase in plant species diversity is beneficial for the quantity and stability of quality corrected yields, i.e. MPP yields, in intensively managed grasslands and across a wide range of pedo-climatic conditions. By expressing these results in terms of DM MPP and MPP yields and their variability, this study is the first which draws the direct link to secondary production and thus to revenue streams from such. The positive economic diversity effect has important implications for farmers' long-term perspective. Firstly, the direct insurance value of plant species diversity reduces temporal risk and thus increases farmers' robustness to shocks.

Secondly, gains from quality corrected yields and revenues due to plant species diversity increase revenues and flexibility and thus contribute to higher adaptability of farm management. It follows that plant species diversity can contribute to overall more resilient and stable agricultural systems (see e.g. Meuwissen et al. 2019).

2.6 Data Availability

The data used in this study is available at Kirwan et al. 2014.

2.7 Code Availability

The Stata code for the econometric estimations as well as the R (R Core Team 2018) code for data preparation are available in the online Appendix.¹⁵

2.8 Acknowledgements

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¹⁵ See Schaub, S., Buchmann, N., Lüscher, A. & Finger, R. (2020). Economic benefits from plant species diversity in intensively managed grasslands. *Ecological Economics*, **168**, 106488. <https://doi.org/10.1016/j.ecolecon.2019.106488>

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2.10 Appendix

2.10.1 Tables

Table A1: Site information.

Mixture type	Site indication	Location	Period of the experiment	Years of yield forage quality data	Years of the experiment	Cuts per year
Mid-European ¹	10	Germany_a	2004-2006	1	2	5
	11	Germany_b	2005-2006	2	1-2	4/4
	15	Ireland_a	2004-2006	2	1,3	5/5
	18	Lithuania_a	2003-2005	3	1-3	3/3/3
	20	Lithuania_c	2004-2006	3	1-3	3/3/2
	22	Norway_a	2004-2006	2	1-2	3/3
	24	Norway_c	2003-2005	2	1-2	3/3
	26	Poland_a	2004-2006	3	1-3	4/4/3
	27	Poland_b	2005-2006	2	1-2	4/3
	35	Wales_a	2003-2006	2	2-3	4/4
North-European ²	36	Wales_b	2004-2006	1	1	4
	13	Iceland_a	2003-2005	3	1-3	2/2/2
	14	Iceland_b	2004-2006	3	1-3	2/2/2
Other ³	23	Norway_b	2003-2006	1	1	2
	1	Belgium	2003-2005	1	1	4
	45	Ireland_d	2010-2011	1	1	5

Classification of clusters and site indication are based on Kirwan et al. (2014). ¹The Mid-European mixture type includes *Lolium perenne*, *Dactylis glomerata*, *Trifolium pratense* and *Trifolium repens*.

²The North-European mixture type includes *Phleum pratense*, *Poa pratensis*, *Trifolium pratense* and *Trifolium repens*. ³The other mixture type includes *Lolium perenne*, *Phleum pratense*, *Trifolium pratense* and *Trifolium repens*.

Table A2: Estimation results of expected outcome and variance of expected outcome (Equations 11 and 13). Results include expected outcome and variance of dry matter yields (DM yields), milk production potentials per kg of DM yield (DM MPP) and milk production potential yields (MPP yields).

	DM yields	Var (DM yields)	DM MPP	Var(DM MPP)	MPP yields	Var (MPP yields)
Intercept	13105.82 (69.83)***	6547610.3 (13.34)***	2.11 (393.56)***	0.004 (1.24)	27514.77 (70.72)***	30662372 (12.69)***
Simpson index ^{0.5} (D ^{0.5})	2599 (6.24)***	-2791192.6 (-2.41)**	-0.02 (-1.31)	-0.011 (-1.16)	5423.78 (6.32)***	-11349391 (-1.95)*
Sown density (Density; 0=low, 1=high)	190.42 (3.19)***	219783.45 (0.88)	-0.01 (-0.7)	0.007 (1.58)	375.45 (2.92)**	757824.46 (0.69)
Year of the experiment (Year)						
2	-836.29 (-0.86)	-1731787.7 (-1.13)	-0.02 (-0.92)	-0.001 (-0.5)	-2055.06 (-0.98)	-8805375.7 (-1.24)
3	-2274.51 (-1.73)	-1784775 (-1.01)	-0.06 (-3.62)***	0.01 (0.86)	-5205.87 (-1.85)*	-7292065.8 (-0.92)
Site (Site)						
10	-1170.28 (-1.24)	-2266710.2 (-1.36)	0.17 (8.52)***	0.002 (1.21)	-310.36 (-0.15)	-11180737 (-1.43)
11	-1492.92 (-3.28)***	-1974259.1 (-2.18)**	0.21 (21.11)***	0.003 (1.91)*	-454.79 (-0.47)	-7660550.8 (-1.78)*
13	-9183.59 (-13.95)***	-3240717.4 (-3.88)***	-0.06 (-5.39)***	0.018 (4.72)***	-19405.67 (-13.72)***	-16064889 (-4.19)***
14	-10096.03 (-15.34)***	247982.79 (0.3)	0.04 (3.35)***	0 (0.04)	-20941.05 (-14.81)***	-486319.72 (-0.13)
15	-398.03 (-0.59)	299600.27 (0.33)	0.09 (10.39)***	-0.005 (-0.82)	51.85 (0.04)	-2268889 (-0.55)
18	-6697.65 (-10.17)***	1026133.2 (1.23)	-0.01 (-1.12)	-0.002 (-0.51)	-14073.46 (-9.95)***	2402855.4 (0.63)
20	-11248.17 (-3.14)***	-5464436.8 (-1.03)	0.2 (5.55)***	-0.037 (-1.08)	-22162.88 (-2.89)**	-24382916 (-1.03)
22	-4031.78 (-8.87)***	3900508.5 (4.31)***	0.1 (9.84)***	0.002 (1.04)	-7500.47 (-7.7)***	17429519 (4.05)***
23	-4452.89 (-59.71)***	-2037647.9 (-9.84)***	-0.05 (-18.58)***	0.001 (0.54)	-9919.71 (-64.57)***	-12766270 (-12.27)***
24	-2931.08 (-5.82)***	-2209741 (-2.81)**	-0.02 (-1.4)	0.002 (2.88)**	-6330.62 (-5.85)***	-11749507 (-3.25)***
26	-9023.92 (-1.91)*	-8038533.9 (-1.13)	0.27 (5.98)***	-0.039 (-0.85)	-17877.46 (-1.77)*	-39619601 (-1.25)
27	-25572.13 (-6.6)***	-2411366.8 (-0.38)	-0.11 (-1.28)	0.183 (32.9)***	-52716.8 (-6.34)***	-2259037.1 (-0.08)
35	-3646.97 (-3.69)***	-702928.68 (-0.56)	0.12 (6.52)***	-0.004 (-0.61)	-6627.04 (-3.12)***	-3873599 (-0.67)
36	-5833.31 (-2.9e+14) ***	-4163286.7 (-3.7e+14) ***	0.07 (1.9e+13) ***	-0.001 (-2.7e+12) ***	-11669.97 (-1.0e+14) ***	-19730794 (-5.3e+13) ***
45	-5010.18 (-130.07)***	-4542317.3 (-25.63)***	0.11 (21.73)***	-0.007 (-2.53)**	-9517.8 (-120.86)***	-20847751 (-24.16)***
Site × Cuts per year (Site×Cuts)						
20	1910.78 (1.64)	1301170.7 (0.71)	-0.07 (-6.75)***	0.013 (1.11)	3480.83 (1.4)	4680635.1 (0.58)
26	928.73 (0.8)	1949525.5 (1.07)	-0.04 (-4.02)***	0.01 (0.89)	1893.56 (0.76)	9008473.3 (1.11)
27	5853.8 (5.99)***	183860.65 (0.12)	0.02 (1.05)	-0.045 (-33.84)***	11953.31 (5.69)***	-2342880.2 (-0.33)
Number of observations	698	698	698	698	698	698
Adjusted R ²	0.766	0.157	0.465	0.025	0.787	0.161

Table includes DM yields, DM MPP and MPP yields. Numbers in parentheses are t-values. * p < 0.1; ** p < 0.05; *** p < 0.01. T-values are corrected for clustered error terms and heteroscedasticity.

Table A3: Estimation results for expected outcome and variance of expected outcome (Equations 11 and 13). Results include expected outcome and variance of metabolizable energy per kg of DM yield (DM ME) and metabolizable energy yields (ME yields).

	DM ME	Var(DM ME)	ME yields	Var(ME yields)
Intercept	10.87 (471.29)***	0.05 (0.55)	142103.81 (70.76)***	799100000 (12.81)***
Simpson index ^{0.5} (D ^{0.5})	-0.06 (-0.94)	-0.23 (-0.92)	28153.8 (6.35)***	-306000000 (-2.05)*
Sown density (Density; 0=low, 1=high)	-0.03 (-0.79)	0.19 (1.56)	1924.59 (2.93)**	20946406 (0.72)
Year of the experiment (Year)				
	-0.07 (-0.82)	-0.01 (-0.3)	-10240.97 (-0.95)	-226600000 (-1.22)
2	-0.26 (-3.66)***	0.3 (0.97)	-26418.83 (-1.83)*	-196100000 (-0.93)
3				
Site (Site)				
	0.67 (7.82)***	0.05 (1.01)	-4343.41 (-0.42)	-287800000 (-1.41)
10	0.83 (19.73)***	0.07 (1.52)	-5551.3 (-1.11)	-210500000 (-1.88)*
11	-0.27 (-5.7)***	0.47 (4.46)***	-100169.49 (-13.83)***	-410500000 (-4.07)***
13	0.2 (4.17)***	-0.04 (-0.41)	-108356.38 (-14.96)***	3479964.6 (0.03)
14	0.38 (10.64)***	-0.14 (-0.85)	-504.1 (-0.07)	-39965016 (-0.37)
15	-0.05 (-1.14)	-0.07 (-0.68)	-72676.81 (-10.03)***	86094466 (0.85)
18	0.8 (4.79)***	-1.03 (-1.1)	-116339.38 (-2.96)***	-649900000 (-1.03)
20	0.39 (9.2)***	0.04 (1.02)	-39921.26 (-7.99)***	464300000 (4.14)***
22	-0.15 (-12.57)***	0.03 (0.67)	-50198.55 (-63.23)***	-315600000 (-11.82)***
23	-0.07 (-1.46)	0.04 (2.56)**	-32534.36 (-5.86)***	-297700000 (-3.13)***
24	1.19 (5.47)***	-1.19 (-0.95)	-93218.01 (-1.8)*	-1017000000 (-1.2)
26	-0.76 (-2.16)**	4.92 (39.83)***	-275139.73 (-6.46)***	-99873246 (-0.13)
27	0.49 (6.8)***	-0.12 (-0.76)	-35261.56 (-3.25)***	-95900942 (-0.63)
35	0.3 (1.3e+14)***	-0.01 (-1.1e+12)***	-60707.94 (-3.7e+13)***	-510900000 (-1.8e+14)***
36	0.48 (19.76)***	-0.16 (-2.02)*	-49891.05 (-122.69)***	-542200000 (-24.3)***
45				
Site × Cuts per year (Site×Cuts)				
	-0.29 (-5.32)***	0.35 (1.12)	18702.6 (1.46)	134500000 (0.62)
20	-0.2 (-3.7)***	0.31 (0.99)	9756.04 (0.76)	234500000 (1.08)
26	0.17 (1.91)*	-1.22 (-42.07)***	62544.38 (5.81)***	-44962444 (-0.24)
27				
Number of observations	698	698	698	698
Adjusted R ²	0.398	0.023	0.783	0.16

Numbers in parentheses are t-values. * p < 0.1; ** p < 0.05; *** p < 0.01. T-values are corrected for clustered error terms and heteroscedasticity.

2.10.2 Figures

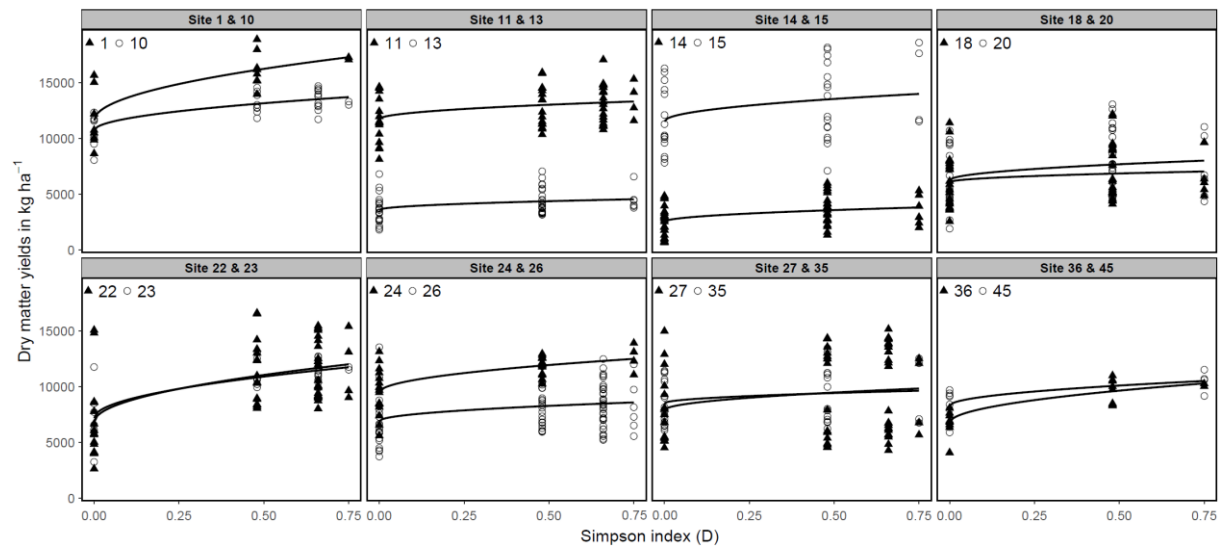


Figure A.1: Dry matter yields in kg ha⁻¹ per site. Note for site 10 that forage quality was only measured for the first four out of five cuts each year. These observations (of the four cuts) are included as this is consistent for all plots and the first cuts are agronomically more important than later ones.

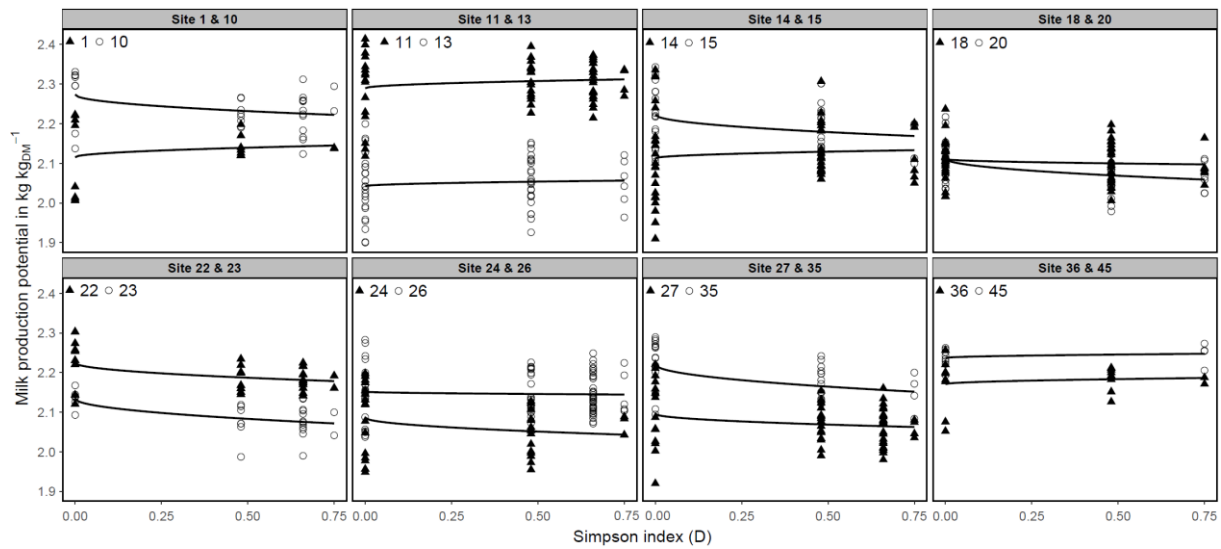


Figure A.2: Milk production potential in kg kg_{DM}⁻¹ per site. Note for site 10 that forage quality was only measured for the first four out of five cuts each year. These observations (of the four cuts) are included as this is consistent for all plots and the first cuts are agronomically more important than later ones.

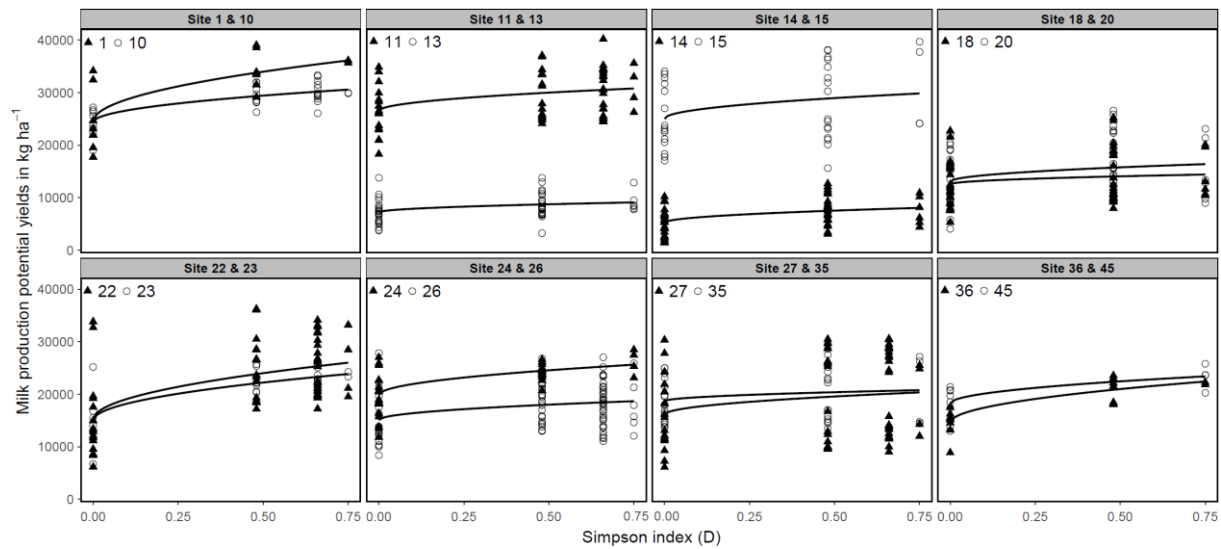


Figure A.3: Milk production potential yields in kg ha⁻¹ per site. Note for site 10 that forage quality was only measured for the first four out of five cuts each year. These observations (of the four cuts) are included as this is consistent for all plots and the first cuts are agronomically more important than later ones.

3 Chapter 3 – Plant Diversity Effects on Forage Quality, Yield and Revenues of Semi-natural Grasslands¹⁶

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Abstract

In agricultural settings, plant diversity is often associated with low biomass yield and forage quality, while biodiversity experiments typically find the opposite. We address this controversy by assessing plant diversity effects on biomass yield, forage quality (i.e. nutritive values), quality-adjusted yield (biomass yield × forage quality) and revenues across different management intensities (extensive to intensive) on subplots of a large-scale grassland biodiversity experiment, using a 1 year study duration. Plant diversity substantially increased quality-adjusted yield and revenues. These findings hold for a wide range of management intensities, i.e. fertilization levels and cutting frequencies, in semi-natural grasslands. Thus, plant diversity was an important production factor independent of management intensity, which was increasing quality-adjusted yield and revenues just like increasing fertilization and cutting frequency. Consequently, maintaining and re-establishing plant diverse grasslands could be a way to sustainably manage temperate grasslands.

Keywords: biodiversity, forage quality, management, Jena Experiment, nutritive value, economic revenues

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3.1 Introduction

Grasslands play a crucial role in global food security and are economically important, as they represent an essential basis for milk and meat production in many regions of the world (Sanderson et al. 2004, Huyghe et al. 2014). Biomass yield, forage quality (i.e. nutritive values) and the resulting quality-adjusted yield (biomass yield \times forage quality) are economically relevant production aspects. Higher plant diversity in agricultural settings is often associated with lower biomass yield and additionally with lower forage quality (Peeters and Janssens 1998, Tallowin and Jefferson 1999, Jefferson and Robertson 2000, Bruinenberg et al. 2002, White et al. 2004, Isselstein et al. 2005), and thus is assumed to have a lower economic value for farmers. This association of higher plant diversity with lower biomass yield and forage quality can be caused by a biased comparison: low diversity swards in agricultural settings are typically the result of high intensity management practices, i.e., based on sown, highly productive species or mixtures (grass-clover) being intensively fertilized, sometimes even on arable land (i.e. intensive or high-input low-diversity systems; sensu Tilman et al. 2006). In contrast, species-diverse (semi-natural) grasslands are often confined to rather unproductive soils and unfavorable climatic conditions. They are typically extensively managed and are nowadays often part of special agri-environmental programs and compensation schemes, which restrict or prohibit fertilization and prescribe late harvests (i.e. extensive or low-input high-diversity systems; sensu Tilman et al. 2006). As a consequence, these diverse swards usually have low annual biomass yield and low forage quality (Barnes et al. 2003, Sanderson et al. 2004).

In contrast to these agricultural settings, plant diversity in biodiversity experiments has been shown to increase biomass yield (Tilman et al. 1996, Hector et al. 1999, Kirwan et al. 2007, Marquard et al. 2009, Finn et al. 2013). This relationship was also confirmed in experiments along a management intensity gradient, i.e. different fertilization levels and/or cutting frequencies (Nyfeler et al. 2009, Weigelt et al. 2009, Vogel et al. 2012, Craven et al. 2016, Yin et al. 2017). However, findings of a plant diversity effect on forage quality (including contents of crude protein, fiber, energy and digestibility) in both single and multiple site experiments are ambiguous, and the effects were often reported to be small (Deak et al. 2007, Nyfeler et al. 2011, Khalsa et al. 2014, Sturludóttir et al. 2014, Ergon et al. 2017, Jing et al. 2017). Important for the productivity of ruminant livestock is the quality-adjusted yield as it represents an integrated measure of biomass yield and forage quality that describes how much quality, for example energy, is available per area. Some studies showed that plant diversity increased quality-adjusted yield, mainly driven by a strong positive effect on biomass yield, also when considering variation in management intensity at a single site (Nyfeler et al. 2011, Oelmann et al. 2015, Suter et al. 2015, Ergon et al. 2017, Schaub et al. 2020). None of these studies considered management intensities from extensive to intensive, a distinct plant diversity gradient from low- to high-diversity systems and a wide range of quality measures at a single site. However, this is required to disentangle the plant diversity

effect on biomass yield, forage quality and quality-adjusted yield from other environmental or management effects, which are always present in agricultural settings.

Furthermore, previous research has shown economic benefits of plant diversity for farmers due to higher revenues and lower production risks when considering biomass yield (Finger and Buchmann 2015) and forage quality (Binder et al. 2018, Schaub et al. 2020). However, there is a lack of evidence on the economic value of plant diversity effects accounting for forage quality and quality-adjusted yields over varying management intensities at a single site, although such assessments could support decision makers in comparing the economic benefits along increasing plant diversity and management intensity.

To resolve the dichotomy between extensive high-diversity systems and intensive low-diversity systems, we propose a conceptual framework considering both biomass yield and forage quality to assess revenues for milk production (Fig. 1). Here, we focus on four contradicting hypotheses that result from the different observations in agricultural and experimental biodiversity settings:

- A) Hypothesis a (Fig. 1a): Biomass yield and forage quality both decrease with increasing plant diversity. Thus, quality-adjusted yield and farm revenues are strongly decreasing with increasing plant diversity.
- B) Hypothesis b (Fig. 1b): Biomass yield increases but forage quality decreases with increasing plant diversity at similar rates. Hence, quality-adjusted yield and revenues remain constant across different levels of plant diversity.
- C) Hypothesis c (Fig. 1c): Biomass yield increases while forage quality stays constant with increasing plant diversity. Hence, quality-adjusted yield and thus revenues increase with increasing plant diversity.
- D) Hypothesis d (Fig. 1d): Biomass yield and forage quality both increase with increasing plant diversity. Thus, quality-adjusted yield and farm revenues are strongly increasing with increasing plant diversity.

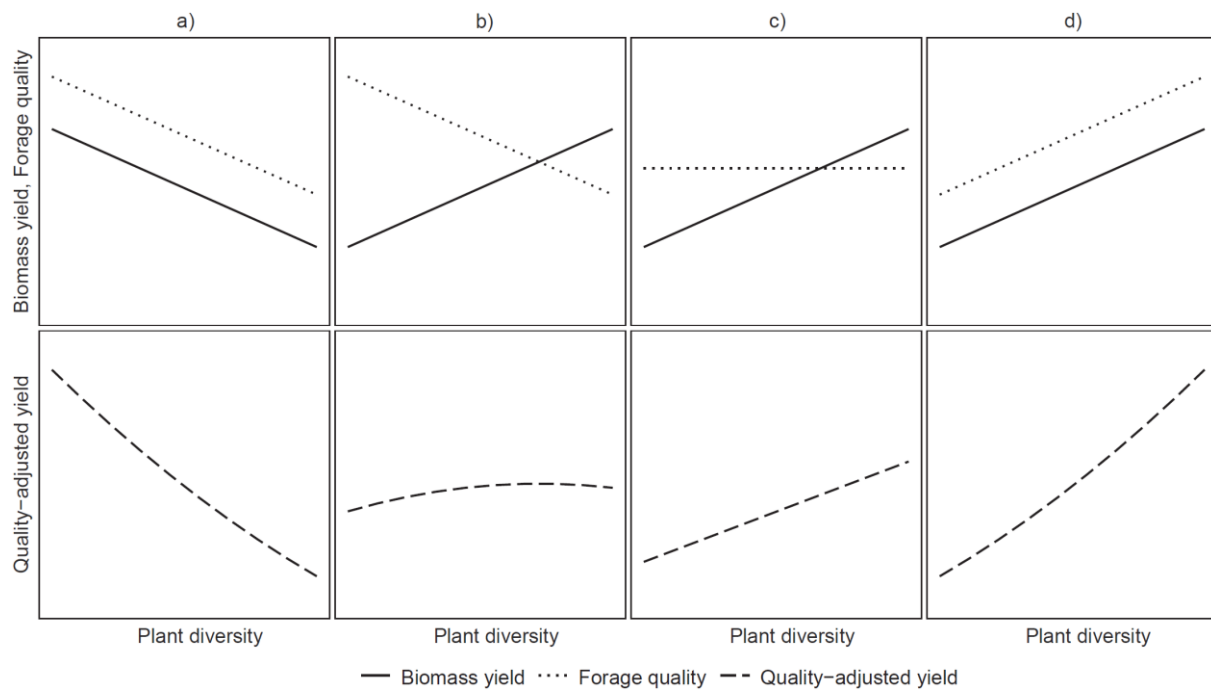


Figure 1: Conceptual framework of the relationship between plant diversity and biomass yield, forage quality and quality-adjusted yield. The panels show the different hypotheses about the plant diversity effect on biomass yield, forage quality and quality-adjusted yield. Panel a: Hypothesis a. Panel b: Hypothesis b. Panel c: Hypothesis c. Panel d: Hypothesis d. Source data are provided as a Source Data file.

To test these hypotheses and to compare plant diversity effects with management effects, a biodiversity experiment with different management intensities was set up within a long-term biodiversity experiment (Jena Experiment; Roscher et al. 2004, Weigelt et al. 2009). This experiment included different plant diversity levels, from 1 to 60 species, and different management intensities, ranging from one cut per year and zero fertilization (extensive) to four cuts and fertilization of 200 kg N ha⁻¹ a⁻¹ (very highly intensive; Weigelt et al. 2009). The three intermediate management intensities are defined as: less intensive, intensive and highly intensive (Fig. A1; Table A1). The experiment also included different levels of legume shares within each plant diversity level, as nitrogen-fixing legumes play an important role for biomass production and forage quality (Spehn et al. 2002, Lüscher et al. 2014). In our study, we measure relevant forage quality variables (cf. Ball et al. 2001; Table A2) and test the effect of plant diversity on biomass yield, different variables of forage quality, quality-adjusted yield, and revenues from potential milk production for different management intensities. We especially focus on metabolizable energy because it is considered a useful measure for overall ruminant-specific nutritional value as it is usually the first limiting factor for ruminant production (Barnes et al. 2003). We also assess milk production potential and revenues because they represent

direct information about animal production and economic implications, which are useful instruments to make better-informed decisions about processes on farm and policy levels.

We find that plant diversity increased quality-adjusted yield and revenues across a wide range of management intensities. Consequently, our findings suggest that maintaining and re-establishing plant diverse grasslands can contribute to sustainable management of temperate grasslands.

3.2 Results

3.2.1 Forage Evaluation

All measures of quality-adjusted yield (i.e. biomass yield \times forage quality), including yields of metabolizable energy ($\text{MJ m}^{-2} \text{a}^{-1}$), milk production potential ($\text{kg m}^{-2} \text{a}^{-1}$), crude protein ($\text{g m}^{-2} \text{a}^{-1}$), utilizable crude protein ($\text{g m}^{-2} \text{a}^{-1}$), organic matter ($\text{g m}^{-2} \text{a}^{-1}$) and neutral detergent fiber ($\text{g m}^{-2} \text{a}^{-1}$) increased significantly with plant diversity, independent of the management intensity (Fig. 2). The only exception was utilizable crude protein yield in the highly intensive management, which was only measured for the first cut of the year. More details about the results, the plant diversity effects on metabolizable energy yield were not significantly different between management intensities (Fig. 3). Increasing the number of species, for example from 1 to 16 (1 to 60), the average predicted metabolizable energy yield of all management intensities increased from 4.1 to 6.6 (9.6) ($\text{MJ m}^{-2} \text{a}^{-1}$). The plant diversity effects on milk production potential yield did also not significantly differ between management intensities (Fig. 3). Increasing the number of plant species from 1 to 16 (60) resulted in an average predicted increase of the milk production potential yield of all management intensities from 0.8 to by 1.2 (1.8) ($\text{kg m}^{-2} \text{a}^{-1}$).

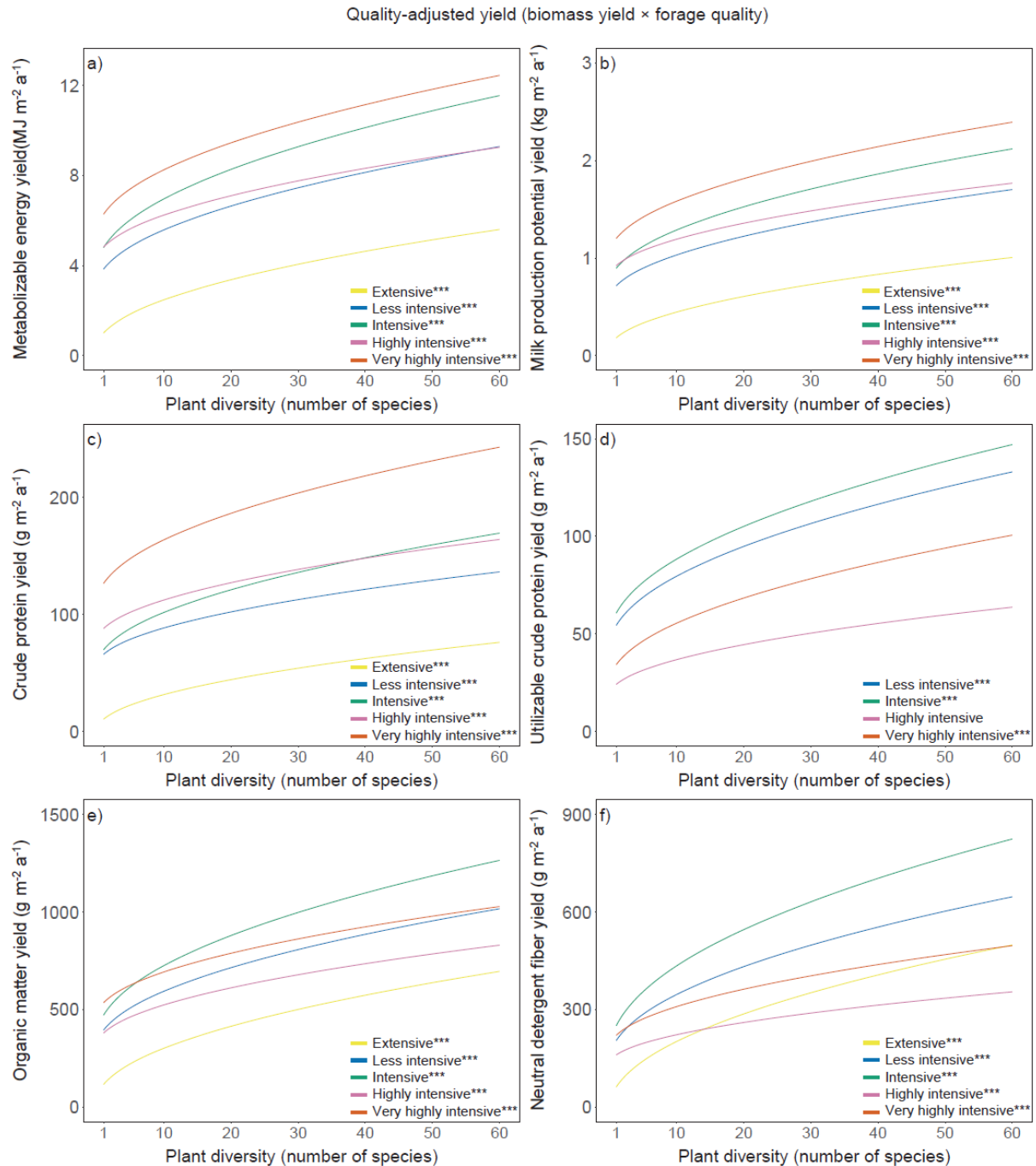


Figure 2: Predicted values of annual quality-adjusted yields (i.e. biomass yield \times forage quality). Predicted annual quality-adjusted yields as a function of plant diversity and management intensity include metabolizable energy yield ($\text{MJ m}^{-2} \text{a}^{-1}$; a), milk production potential yield ($\text{kg m}^{-2} \text{a}^{-1}$; b), crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$; c), utilizable crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$; d), organic matter yield ($\text{g m}^{-2} \text{a}^{-1}$; e) and neutral detergent fiber yield ($\text{g m}^{-2} \text{a}^{-1}$; f). *, **, *** denote significance at the 5%, 1% and 0.1% level of the plant diversity effect per management intensity, respectively (corrected for multiple comparisons). The significance levels are based on a mixed effect model, see Equation (4). The corresponding coefficients for the plant diversity effect per management intensity can be found in Table A3 to A5. Utilizable crude protein content was only measured for the first cut of the year.

Effect	Variable	extensive – less intensive	extensive – intensive	extensive – highly intensive	extensive – very highly intensive	less intensive – intensive	less intensive – highly intensive	less intensive – very highly intensive	intensive – highly intensive	intensive – very highly intensive	highly intensive – very highly intensive
a) Plant diversity effect per management intensity	Metabolizable energy yield	-0.13	-0.32	0.02	-0.23	-0.19	0.15	-0.11	0.34	0.09	-0.26
	Milk production potential yield	-0.02	-0.06	-0.003	-0.05	-0.04	0.02	-0.03	0.06	0.005	-0.05
	Crude protein yield	-0.71	-5.04	-1.55	-7.53	-4.32	-0.84	-6.81	3.49	-2.49	-5.98
	Organic matter yield	-6.28	-31.73	19.15	12.85	-25.45	25.43	19.13	50.88	44.58	-6.3
	Neutral detergent fiber yield	-0.83	-20.42	35.97	23.83	-19.59	36.8	24.66	56.38	44.25	-12.14
	Biomass yield	-7.43	-35.63	18.41	10.99	-28.19	25.84	18.42	54.03	46.61	-7.42
b) Management effect	Metabolizable energy yield	-1.2	-3.1	-3.4	-5.5	-1.9	-2.2	-4.3	-0.3	-2.4	-2.1
	Milk production potential yield	-0.2	-0.6	-0.7	-1	-0.4	-0.4	-0.8	-0.1	-0.5	-0.4
	Crude protein yield	-19.3	-38.9	-65.9	-109.9	-19.7	-46.6	-90.6	-27	-71	-44
	Organic matter yield	-131.3	-308.9	-277.2	-479.8	-177.6	-145.9	-348.5	31.7	-170.9	-202.6
	Neutral detergent fiber yield	-71.9	-176.1	-138.3	-226.3	-104.2	-66.4	-154.4	37.8	-50.2	-88
	Biomass yield	-148.3	-343.3	-318.7	-543.3	-195	-170.4	-395	24.6	-200	-224.6

Figure 3: Differences between plant diversity effects per management intensity and management effects. The figure shows differences and significance of the differences between a) plant diversity effects per management intensity and b) management effects on quality-adjusted yield (g or MJ or kg m⁻² a⁻¹) and biomass yield (g m⁻² a⁻¹). Pairs of management intensities indicate the management intensities compared to each other. The displayed numbers are the differences between the effects of the compared management intensities. Light blue, mid blue, dark blue denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons), and white denotes no significant effect at the 5% level. The significance levels are based on the Wald test. Note that utilizable crude protein content was only measured for the first cut of the year, which is why we do not compare it between management intensities. The figure is based on Table A3 to A8.

Two factors underlie the plant diversity effect on quality-adjusted yield, namely effects on biomass yield and on forage quality. First, we found a positive relationship between plant diversity and biomass yield (g m⁻² a⁻¹), which was robust across all management intensities (Fig. 4). This effect was highest for the intensive management. The difference in the plant diversity effect between this intensive

management and the others was significant for all but the extensive management. The plant diversity effects on biomass yield of the other intensities were not significantly different from each other (Fig. 3). Second, the plant diversity effect on forage quality differed among forage quality variables, but the effects were small and insignificant in most of the cases (Fig. 5). More specifically, plant diversity had no effect on metabolizable energy content ($\text{MJ kg}^{-1} \text{a}^{-1}$) and milk production potential ($\text{kg kg}^{-1} \text{a}^{-1}$), except a significant and slightly negative effect in the intensive management. When increasing plant diversity from 1 to 16 (60) species, these relative predicted effects for the intensive management on metabolizable energy content and on milk production potential were -4.1% (-9.2%) and -4.9% (-11.1%), respectively. The plant diversity effects on quality-adjusted yield, biomass yield and forage quality were robust when controlling for legume share instead of legume presence in the analysis (Table A9 to A14).

Generally, the management effect on quality-adjusted yield and biomass yield increased from extensive to very highly intensive management, except for intensive and highly intensive management which did not differ significantly from each other (Fig. 2-4). The management effect on forage quality was generally as expected, i.e. higher management intensity increased energy and protein contents (Table A3 and A4).

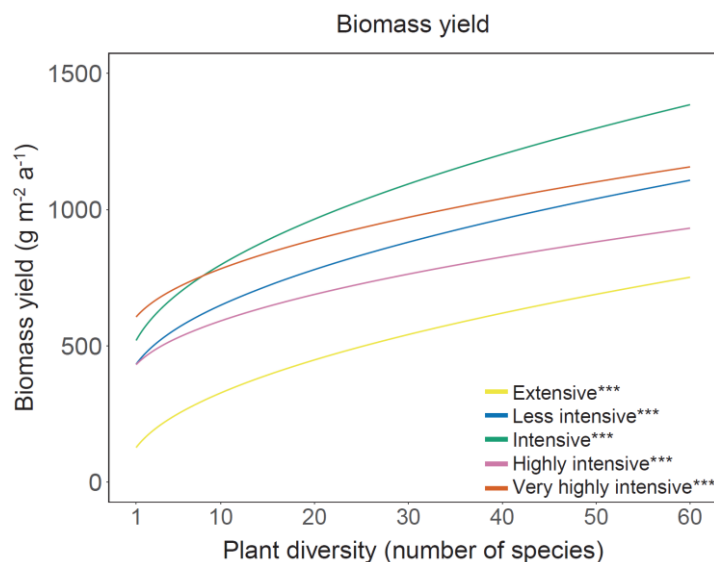


Figure 4: Predicted annual biomass yield ($\text{g m}^{-2} \text{a}^{-1}$) as a function of plant diversity and management intensity. *, **, *** denote significance at the 5%, 1% and 0.1% level of the plant diversity effect per management intensity, respectively. The significance levels are based on a mixed effect model, see Equation (4). The corresponding coefficients for the plant diversity effect per management intensity can be found in Table A6.

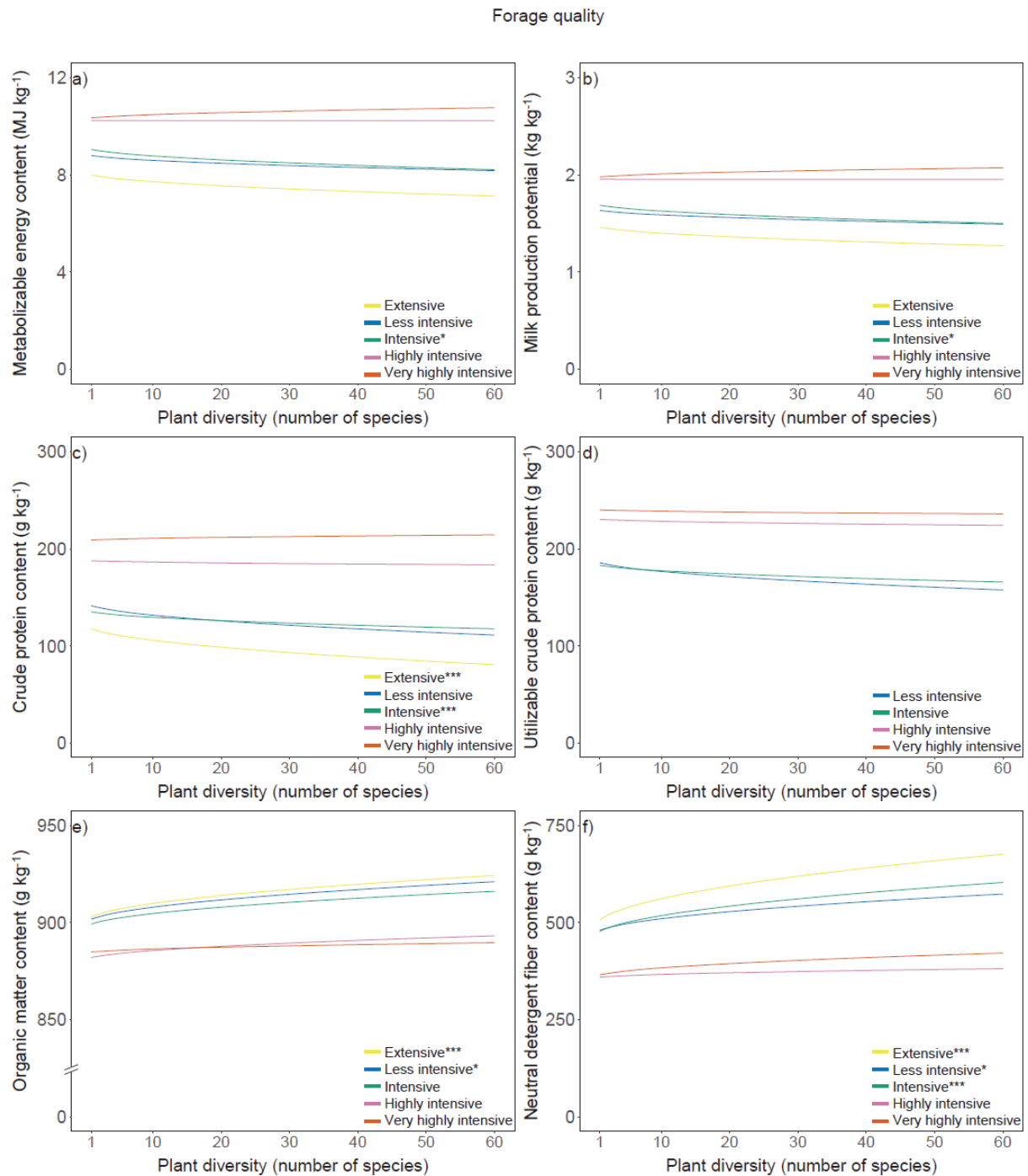


Figure 5: Predicted values of annual average forage quality. Predicted annual average forage quality as a function of plant diversity levels and management intensity include metabolizable energy content (MJ kg⁻¹; a), milk production potential (kg kg⁻¹; b), crude protein content (g kg⁻¹; c), utilizable crude protein content (g kg⁻¹; d), organic matter content (g kg⁻¹; e) and neutral detergent fiber content (g kg⁻¹; f). *, **, *** denote significance at the 5%, 1% and 0.1% level of the plant diversity effect per management intensity, respectively (corrected for multiple comparisons). The significance levels are based on a mixed effect model, see Equation (4). The corresponding coefficients for the plant diversity effect per management intensity can be found in Table A3 to A5. Utilizable crude protein content was only measured for the first cut of the year. Note that the y axis break in the figure of organic matter content.

3.2.2 Economic Valuation

Our results reveal a positive relationship between plant diversity and economic performance, here expressed as revenues from potential milk production (Euro ha⁻¹ a⁻¹; Fig. 6). This finding was independent of the management intensity. The positive impact of plant diversity on revenues was not significantly different between management intensities (Table A8). On average across all management intensities, an increase of plant diversity from 1 to 16 (1 to 60) plant species increases revenues by about +1400 Euro ha⁻¹ a⁻¹ (+3100 Euro ha⁻¹ a⁻¹). Overall, the management effect increased from the extensive to the very highly intensive management, yet with no differences between the intensive and the highly intensive management. It is noteworthy that the management effect of the extensive management was by far the lowest compared to all other management intensities. In economic terms, the predicted change in revenues due to switching from less intensive to intensive was +550 Euro ha⁻¹ a⁻¹; and +1500 Euro ha⁻¹ a⁻¹ if switching from less intensive to the very highly intensive management (effect of size *a* and *b* respectively in Fig. 6). These effect sizes were of about the same magnitude as changing plant diversity in the less intensive management from 1 to 5 or from 1 to 19 species, respectively.

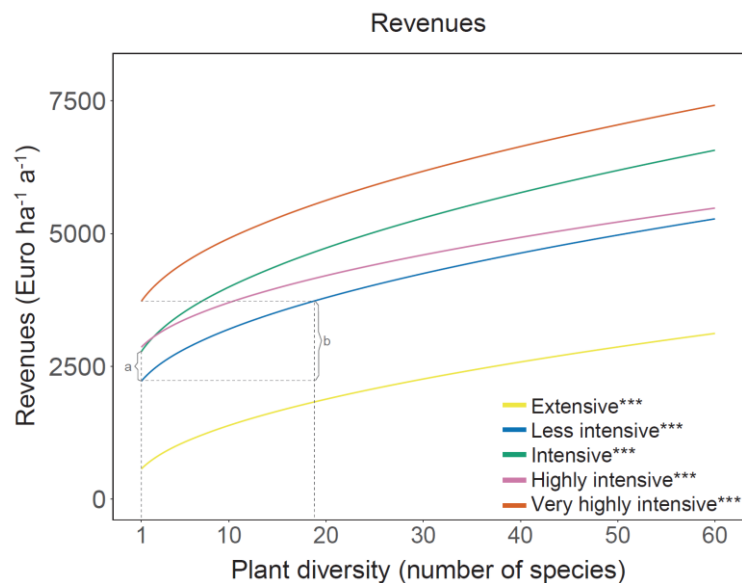


Figure 6: Predicted revenues (Euro ha⁻¹ a⁻¹) as a function of plant diversity and management intensity. *a* indicates the effect size of changing management intensity from less intensive to intensive or changing plant diversity in the less intensive management from 1 to 5 species. *b* indicates the effect size of changing the management intensity from less intensive to the very highly intensive or changing plant diversity in the less intensive management from 1 to 19 species. *, **, *** denote significance at the 5%, 1% and 0.1% level of the plant diversity effect per management intensity, respectively (corrected for multiple comparisons). The significance levels are based on a mixed effect model, see Equation (4). The corresponding coefficients for the plant diversity effect per management intensity can be found in Table A6.

3.3 Discussion

We presented an analysis of plant diversity effects on different measures of quality-adjusted yield in temperate semi-natural grasslands exposed to different management intensities and we quantified the economic implications of plant diversity. The results show that plant diversity increases quality-adjusted yield by increasing biomass yield at rather constant forage quality. While this does not confirm one individual of our four distinct hypotheses, it supports the notion of hypothesis c (Fig. 1) and nuanced forms of it, i.e. that a strong positive plant diversity effect on biomass yields at rather constant forage quality leads to higher quality-adjusted yield.

The observation of an overall positive and robust influence of plant diversity on quality-adjusted yield, especially based on biomass effects (Table A15), is consistent with findings from other experiments that investigated yields of crude protein and energy (Nyfeler et al. 2011, Oelmann et al. 2015, Suter et al. 2015, Ergon et al. 2017, Schaub et al. 2020). Further confirmation is given by studies that only reported biomass yield and forage quality but not quality-adjusted yield (Deak et al. 2007, Khalsa et al. 2014, Sturludóttir et al. 2014, Jing et al. 2017).

The positive plant diversity effect on biomass yield has been related to complementarity and selection effects (Tilman et al. 1997, Loreau and Hector 2001; but see Barry et al. 2019). The complementarity effect is assumed to be caused by resource partitioning of different species or positive species interactions while the selection effect is supposed to be due to a higher probability of including highly productive species in more plant diverse grasslands. For the Jena Experiment, both the complementarity and the selection effects contributed to higher biomass yield with increasing diversity, with the complementarity effect getting stronger and the selection effect getting weaker over time (Marquard et al. 2009, Weisser et al. 2017; see Weigelt et al. (2009) for detailed discussion of biomass yield data).

Furthermore, our analysis demonstrated that strong plant diversity effects on biomass yield in some management intensities were partially counterbalanced by declines in metabolizable energy content and milk production potential. This results in more similar plant diversity effects on metabolizable energy yield and milk production potential yield across all management intensities. This finding is highly relevant when evaluating plant diversity effects on biomass yield across management intensities for agricultural reasons as it implies that plant diversity might be as important for more intensively managed grasslands as for less intensively managed ones. Ensuring high plant diversity in fertilized grasslands is possible with moderate fertilization, but becomes difficult at very high fertilization levels; however, the species loss in our experiment was slow enough to maintain a distinct plant diversity gradient (Weigelt et al. 2009). Moreover, although many grasslands are permanent, there is a high share of grasslands that are frequently over-sown or restored (Neuens et al. 2002, Schils et al. 2007, Creighton et al. 2011, Lesschen et al. 2014, Eurostat et al. 2019). Furthermore, we note that other

studies of the same experiment at later years (Weigelt et al. 2009, Vogel et al. 2012, Oelmann et al. 2015) support our findings and show robustness of these results also consider longer time horizon. Furthermore, we also found that plant diversity can achieve gains in increasing metabolizable energy yield and milk production potential yield similar to increasing fertilization levels and/or cutting frequencies. Hence, the production factor plant diversity can reduce other inputs in semi-natural grasslands while maintaining the same grassland productivity.

Grasslands have important economic, ecological and cultural functions (Sala and Paruelo 1997, Fraser and Chisholm 2000, Sanderson et al. 2004, Huyghe et al. 2014). The management of these grasslands is becoming more complex, as the traditional interests of production and profit optimization are increasingly complemented by concerns about sustainability and provision of ecosystem services and functions (Sanderson et al. 2004, Baulcombe et al. 2009, Power et al. 2010). Previous studies showed the positive impact of plant diversity on many of these ecosystem services and functions in grasslands (De Deyn et al. 2004, Potts et al. 2009, Lindemann-Matthies et al. 2010, Isbell et al. 2011, Connolly et al. 2018, Meyer et al. 2018). In our study we found additionally substantial economic benefits, in terms of revenues, from higher plant diversity. These findings are consistent with previous findings from semi-natural and intensively managed experimental grasslands (Finger and Buchmann 2015, Binder et al. 2018, Schaub et al. 2020). Our results also show that the plant diversity effect between management intensities becomes more similar when forage quality is included in the economic assessment. This implies that plant diversity in more intensively managed grasslands can be as important as in more extensively managed grasslands. Furthermore, according to our findings, increasing plant diversity in semi-natural grasslands can have equally large positive effects on revenues as increasing management intensity. For example, changing the management intensity from less intensive to intensive (very highly intensive) has the same economic benefit as increasing plant diversity from 1 to 5 (19) plant species in the less intensively managed grassland, namely an effect of about +550 Euro ha⁻¹ a⁻¹ or +25% in relative terms (+1500 Euro ha⁻¹ a⁻¹ or +70% in relative terms) on predicted revenues. However, increasing the management intensity would cause additional variable costs for cutting (including labor and fuel) and fertilization, in our example, i.e. changing management intensity from less intensive to intensive (very highly intensive) management, these costs would be about -174 (-493) Euro ha⁻¹ a⁻¹ (see Material and Methods for calculations). However, we did not consider the costs of seeds of species planted in the Jena Experiment, which can be high for species, which are rarely used in agricultural settings. Similarly, prices for diverse mixtures, which are 'ready to sow', tend to be expensive. Taking an example of a German seed provider and assuming reseeding of grasslands using available seed mixtures (20 kg ha⁻¹), prices of highly diverse mixtures (28 to 49 species) are considerably higher (mean of 1203 Euro ha⁻¹; SD = 521) than that of standard mixtures (1 to 8 species; mean of 229 Euro ha⁻¹; adjusted price; SD = 36). Alternatively, fresh hay transfer, i.e.

transferring fresh seed-containing hay from plant diverse grasslands to improve species poor or restore plant diverse grasslands (see e.g. Kiehl et al. 2010), represents a near-natural method and a more cost friendly option, with variable costs of 427 Euro ha⁻¹ (including variable fuel, labor and opportunity costs). However, these costs are depending on the circumstances (see Material and Methods for details). Therefore, restoring methods aiming at increasing plant diversity can be beneficial for farmers when comparing costs to revenue benefits. Furthermore, less fertilizer application would allow maintaining high plant diversity over longer time (Mountford et al. 1993, Gough et al. 2000), and in turn maintaining the plant diversity effect. Moreover, any reduction in management can also entail other ecosystem benefits, such as increasing whole-ecosystem biodiversity (beside solely plant diversity) or decreasing greenhouse gas emissions (Allan et al. 2014, Hörtnagl et al. 2018). Considering all our findings, altering plant diversity even in more intensively used grasslands appears to be a valuable management tool to farmers.

3.4 Conclusion

Our findings suggest that increasing plant diversity in semi-natural grasslands presents a viable strategy for sustainable intensification. Plant diversity represents a production factor to increase quality-adjusted yield independent of management intensity. In this respect, plant diversity was found to be as valuable as increasing the fertilization level and cutting frequency in semi-natural grassland. Therefore, we propose that plant diversity should be considered in farm management decisions and in the design of agri-environmental schemes. The challenge remains to develop management systems using mixtures that allow maintaining a high plant diversity also in fertilized grasslands. Here, pathways to exploit the positive plant diversity effects over longer periods could include increasing livestock diversity to promote plant diversity (Wang et al. 2019), maintaining and promoting species diverse hay meadows, e.g. *Arrhenatheretum elatioris*, with two to three cuts and low to moderate fertilization levels (Dierschke and Briemle 2002), and seeding of plant diverse mixtures containing complementary species and legumes. Such plant diverse mixtures can also be helpful in dealing with droughts (Isbell et al. 2015, Hofer et al. 2016, Haughey et al. 2018), which are becoming more severe and frequent under changing climatic conditions (IPCC 2013). Maintaining and re-establishing plant diverse grasslands could provide a win-win situation as it enables a sustainable increase in quality-adjusted yield and revenues while at the same time it supports other important ecosystem services and functions.

3.5 Methods

3.5.1 Experimental Design

Our study is part of the Jena Experiment, a large-scale and long-term biodiversity-ecosystem functioning experiment in Jena (Thuringia, Germany, 50°55' N, 11°35' E, 130 m a.s.l.; mean annual air temperature 9.9 °C, annual precipitation 610 mm; 1980–2010 (Hoffmann et al. 2014)).

The experimental communities were established in May 2002, covering different plant diversity levels (including 1, 2, 4, 8, 16 and 60 species) and a functional group gradient (including 1, 2, 3 and 4 functional groups) per plot, and were seeded in 82 main plots of a size of 20 × 20 m, adopting a replacement series design. The species pool consists of 60 species typical to Central European *Arrhenatherum* meadows. Species were categorized into four functional groups, grasses (16 species), small herbs (12), tall herbs (20), and legumes (12) using cluster analysis based on an ecological and morphological trait matrix (Roscher et al. 2004). The mixtures were assembled by random selection with replacement, yielding 16 replicates for mixtures with 1, 2, 4 and 8 species and 14 replicates for the 16-species mixtures. In addition, all 60 species were sown on four plots which were used for comparison in the present study. Plots were arranged in four blocks, regularly weeded to maintain the sown plant diversity levels. No fertilization was carried out in the main plots.

The results presented here are from the Management Experiment set up within the Jena Experiment. For the Management Experiment four subplots of 1.6 × 4 m were established in April 2005 within each of the 20 × 20 m main plots. Each subplot represented one of four additional management intensities with varying fertilization level and cutting frequency per year as listed in Table A1 (Weigelt et al. 2009). The core area of the 20 × 20 m main plots served as one management intensity with two cuts per year and zero fertilizer application (less intensive). The five management intensities ranged from extensive to very highly intensive management, including an extensive management (one cut per year, no fertilization) and a very highly intensive management (four cuts per year, high fertilization of 200 kg N ha⁻¹ a⁻¹ and corresponding P and K fertilization, see below) and three intermediate management intensities: less intensive management (two cuts per year, no fertilization), intensive management (two cuts per year, intermediate fertilization of 100 kg N ha⁻¹ a⁻¹) and highly intensive management (four cuts per year, intermediate fertilization of 100 kg N ha⁻¹ a⁻¹). Thus, the experiment consisted of 390 subplots (82 × 4 management subplots plus 82 core areas). We randomized the allocation of management intensities to subplots, except for the extensive subplots, which were always placed at the plot margins due to logistical constraints. The management intensities selected are representative for common grassland management intensities on floodplains comparable to the experimental site, ranging from grasslands in agri-environmental schemes to intensively managed grasslands (Weigelt et al. 2009). We avoided a full factorial design with all fertilization levels per cutting frequency because

such a design would include factor combinations that are not reasonable for agricultural practice such as frequent cutting without fertilization. The controlled manipulation in the experiment of the grassland with different management intensities and different levels of plant diversity, allowed us to test for the presence or absence of a plant diversity effect for different management intensities (Schmid et al. 2002).

For the preparation of the *Management Experiment*, we fertilized all four subplots dedicated to the experiment once with 50 kg N ha⁻¹ a⁻¹, 31 kg P₂O₅ ha⁻¹ a⁻¹, 31 kg K₂O ha⁻¹ a⁻¹, and 2.75 kg MgO ha⁻¹ a⁻¹ in April 2005. Starting in 2006, the fertilized subplots received commercial NPK pellets using a lawn fertilizer distributor in amounts presented in Table A1. The fertilizer was applied in two equal portions: first in early spring (beginning of April) and second after either the first or second cut (respectively for treatments with two or four cuts) in late June. Plots were cut either once, twice or four times during the growing season (Table A16) with sickle bar mowers at approximately 3 cm above ground level. All cut material was removed from the plots. Cutting, fertilizing and weeding were done on a per-block basis such that any maintenance effect was corrected for by the block effect in the statistical analysis.

3.5.2 Data Collection & Laboratory Analysis

We measured biomass yield and several common and relevant forage quality variables in the harvests of 2007 (Table A2; Ball et al. 2001, Barnes et al. 2003). Moreover, we estimated contents of metabolizable energy, (metabolically) utilizable crude protein and milk production potential, all providing valuable information on forage quality related to an agricultural economic perspective.

To measure standing aboveground biomass, we cut all plants within one randomly selected 0.2 × 0.5 m frame in each subplot at 3 cm above ground level, shortly before mowing the rest of a subplot. In the main plots, we cut all plants within four randomly selected 0.2 × 0.5 m frames at 3 cm above ground level. We oven-dried (70°C, 48 h) and weighed all harvested biomass of sown species. Subsequently, we milled the samples to pass a 1-mm sieve (rotor mill type SM1, RETSCH, Haan, Germany). Dry matter and total ash content were analyzed in these samples by drying at 105°C and 550°C, respectively (AOAC index no. 942.05; AOAC, 1997), with a thermogravimetric determinator furnace (TGA 500, LECO Co., St. Joseph, USA). Organic matter was calculated as dry matter minus total ash. Crude protein content was quantified as 6.25 × nitrogen content using a C/N analyzer (Leco-Analysator Typ FP-2000, Leco Instrumente GmbH, Kirchheim, Germany; AOAC index no. 977.02). Following the procedures of Van Soest et al. (1991) using the Fibertec apparatus (Fibertec System M, Tecator, 1020 Hot Extraction, Flawil, Switzerland), we analyzed neutral detergent fiber content, corrected for ash content by addition of sodium sulfite. Contents of ether extract were analyzed with a Soxhlet extractor (Extraktionssystem B-811, Büchi, Flawil, Switzerland; AOAC index no. 963.15). Ether extract content was not reported individually because it is of lower importance as a single variable for forage quality since it represents

only a small share of dry matter, and thus, only of small proportion of energy supply (Van Soest 1994, Beever et al. 2000), but this information was needed for estimating contents of metabolizable and net energy. The content of metabolizable energy was estimated by *in vitro* fermentation with rumen fluid of a dairy cow, applying the Hohenheim Gas Test procedure (Menke and Steingass 1988). In this approach, 200 mg of feed samples were incubated together with 10 mL of rumen fluid and 20 mL of McDougall buffer for 24 hours in glass syringes at 39°C. Afterwards, the gas production was measured by a calibrated scale. Together with compositional information, the content of metabolizable energy was calculated by: metabolizable energy (MJ kg⁻¹) = 3.16 + 0.0695 × fermentation gas (mL day⁻¹) + 0.000730 × fermentation gas (mL day⁻¹)² + 0.00732 × crude protein (g kg⁻¹) + 0.02052 × ether extract (g kg⁻¹). Moreover, we estimated the content of net energy for lactation based on the same system, by: net energy (MJ kg⁻¹) = 1.64 + 0.0269 × fermentation gas (mL day⁻¹) + 0.00078 × fermentation gas (mL day⁻¹)² + 0.0051 × crude protein (g kg⁻¹) + 0.01325 × ether extract (g kg⁻¹). We subsequently used net energy for lactation to estimate the milk production potential of the biomass yield (Jans et al. 2015). In order to estimate utilizable crude protein content, we collected the mixture of rumen fluid and buffer remaining after incubation of the samples in Falcon tubes and analyzed them for ammonium nitrogen content with the Kjeldahl principle using the distillation unit 323 of Büchi Labortechnik AG (Flawil, Switzerland). Utilizable crude protein content was then calculated by the equation described by Edmunds et al. (2012) based on analyzed ammonium content in the incubation fluid, obtained with the Hohenheim Gas test procedure without (blank) and with feed (sample), using an ammonia selective electrode (Metrohm AG, Herisau, Switzerland), and the nitrogen content of the biomass yield samples: utilizable crude protein (g kg⁻¹) = [(NH₃-N_{blank} + N_{sample} - NH₃-N_{sample}) / dry matter (mg)] × 6.25 × 1000.

We performed the chemical analyses for the first and the last cut of the year 2007, except for utilizable crude protein content, which we only estimated for the first cut. To retrieve information about the forage quality of harvests from management intensities that included more than two cuts, we used linear interpolation. This was possible, as the forage quality variables showed either a continuous decrease (metabolizable energy content, milk production potential, crude protein content) or increase (neutral detergent fiber content, ash content) from the first to the last cut. Further, we deleted data from swards that had very small biomass yield or with missing biomass yield information for at least one cut of the year (these were in total 54 swards, from which 31 had a plant diversity level of one species, 11 had a plant diversity level of 2 species, 4 had a plant diversity level of 4 species, 7 had a plant diversity level of 8 species, 3 had a plant diversity level of 16 species, 1 had a plant diversity level of 60 species).

Finally, we calculated the sum of biomass yield of all cuts of a year, i.e. annual biomass yield (g m⁻² a⁻¹), average forage quality of all cuts of a year, i.e. annual average forage quality (g or MJ or kg kg⁻¹), and

the sum of quality-adjusted yield of all cuts of a year, i.e. annual quality-adjusted yield (g or MJ or kg m⁻² a⁻¹):

$$\text{Biomass yield}_h = \sum_{\text{cut}=1}^h \text{Biomass yield}_{\text{cut}} \quad (1)$$

$$\text{Quality}_h = \sum_{\text{cut}=1}^h \text{Quality}_{\text{cut}} \times \frac{\text{Biomass yield}_{\text{cut}}}{\text{Biomass yield}_h} \quad (2)$$

$$\text{Quality-adjusted yield}_h = \sum_{\text{cut}=1}^h \text{Biomass yield}_{\text{cut}} \times \text{Quality}_{\text{cut}} \quad (3)$$

h includes all cuts of a year.

3.5.3 Analysis of the Plant Diversity Effect

We analyzed the plant diversity effect on biomass yield, forage quality and quality-adjusted yield using a mixed effect model:

$$y = \alpha + \beta_{D \times M} D^{0.5} x M + \beta_M M + \beta_{L \times M} L x M + \beta_{FG} FG + \beta_G G + \beta_H H + u_B B + u_P P + e \quad (4)$$

In the Equation (4), the dependent variable was either annual biomass yield, average annual forage quality or annual quality-adjusted yield. To model the effect of plant diversity, the square root specification ($D^{0.5}$) was chosen over others (linear, linear and squared, logarithmic and D^{-1}), as this specification allowed a diminishing plant diversity effect, which is often observed (see e.g. Hooper et al. 2005), and it performed best across the different outcome variables in terms of the Akaike information criterion (AIC) and Bayesian information criterion (BIC). More specifically, we modeled the plant diversity effect for each management intensity by introducing an interaction term of the square root of plant diversity and management intensity ($D^{0.5} \times M$). Moreover, we included the different management intensities by a dummy variable for each management intensity (M), the interaction term of the presence of legumes with management intensities ($L \times M$), number of functional groups (FG), fixed effects for the presence grasses (G) and tall herbs (H) as well as random effects for blocks (B) and plots (P). Finally, we corrected the results for heteroscedasticity by using robust standard errors. In addition, we conducted a robustness analysis, by using a model with the square root of legume share (number of legumes divided by number of all species) instead of the presence of legumes. This allowed us to account in different ways for the importance of legumes (Lüscher et al. 2014) and the possibility that the legume share drives the plant diversity effect on nutritive values. Moreover, based on earlier experiments, the Jena Experiment design paid special attention to the role of legumes in grasslands and the interaction with the plant diversity effect, by including legumes in all plant diversity levels, thus avoiding a confounding effect between plant diversity and presence of legumes. However, not every plot included legumes, which offers the possibility to analyze effects of legume presence or abundance (Roscher et al. 2004).

We used the Bonferroni correction to correct for perform multiple comparisons (=significance levels/n, n equaled number of different forage qualities, i.e. six, except for biomass yield, for which n was one). Furthermore, we tested whether plant diversity effects per management intensity differed from each other by using a Wald test. To conduct the entire data analyses we employed Stata 15.0 for Windows.

3.5.4 Economic Valuation

To evaluate the on-farm value of plant diversity we computed the annual revenues of milk sales:

$$\text{Revenues } (M,D) = \text{Milk production potential yield } (M, D) \times \text{milk price} \quad (5)$$

where milk production potential yield refers to the predicted annual mean milk production potential yield per management intensity for different levels of plant diversity. By using milk revenues we explicitly considered forage quality. The reference price used for the valuation of the milk production potential yield was the average milk price of 2016 and 2017 in Germany of 0.31 Euro kg⁻¹ (BMEL 2018). To adapt dimensions of milk production potential yield to more reasonable dimensions from an agricultural economic perspective, we converted the units of milk production potential yield from kg m⁻² a⁻¹ to kg ha⁻¹ a⁻¹. It shall be emphasized that we assume that farmers maximize their utility, leading to economically efficient decisions (sensu economics). To identify the plant diversity effect we used the same model as described in Equation (4) and the Welch t-test.

Typical variable fertilizer and cutting costs (including fertilizer, labor and fuel costs) in Germany were derived from KTBL (2019), aiming to represent costs in agricultural settings. Costs of increasing fertilization level of our management intensities by one (Table A1) of 165 Euro ha⁻¹ a⁻¹ were computed by the amount of calcium ammonium nitrate and PK fertilizer required to meet the change in N, P and K fertilization multiplied with the respective price (100 N ha⁻¹/0.27 N kg⁻¹ × 0.23 Euro kg⁻¹ + max{43.6 P ha⁻¹/0.12 P kg⁻¹, 83 K ha⁻¹/0.24 K kg⁻¹} × 0.22 Euro kg⁻¹). Additionally, when farmers switch from zero fertilization to some fertilization, variable costs for the process of applying fertilizer (including labor and fuel costs) of 9 Euro ha⁻¹ a⁻¹ arise (0.55 h ha⁻¹ × 13 Euro h⁻¹ + 1.9 l ha⁻¹ × 0.75 Euro l⁻¹). We computed costs of increasing cutting frequency by one cut of 77 Euro ha⁻¹ a⁻¹ considering labor costs and fuel costs for cutting, windrowing and collecting the harvest ((0.67 h ha⁻¹ + 0.53 h ha⁻¹ + 3.56 h ha⁻¹) × 13 Euro h⁻¹ + (4.85 l ha⁻¹ + 3.18 l ha⁻¹ + 12.67 l ha⁻¹) × 0.75 Euro l⁻¹). Furthermore, we included costs of two alternatives for increasing species diversity: reseeding with seed mixtures and fresh hay transfer. The variable costs of reseeding with mixtures comprise two parts, the actual process (reseeding and rolling (KTBL 2019)) and the purchase of the mixture. The process costs (including labor and fuel costs) of 12 Euro ha⁻¹ a⁻¹ are taken from KTBL (2019) for sites 5 km away from the farm ((0.27 h ha⁻¹ + 0.41 h ha⁻¹) × 13 Euro h⁻¹ + (2.08 l ha⁻¹ + 2.46 l ha⁻¹) × 0.75 Euro l⁻¹). For the mixture costs, we collected prices for mixtures from two online retailers (Fig. 7), of which one focuses on highly diverse regional mixtures (highly diverse mixtures; green cycles). We accounted for production cost differences of single seeds

between the shops by replicating the mixtures sold at the standard mixture shop (standard mixtures; orange triangles) with seeds of the shop focusing on species diverse regional mixtures (price-adjusted standard mixtures; green triangles). The species number of the standard and price-adjusted standard mixtures ranges from 1 to 8 and the respective mean (standard deviation) of the prices for reseeded (20 kg ha⁻¹) are 104 (26) and 229 (36) Euro ha⁻¹ a⁻¹. The diverse mixtures include 38 to 49 species and the mean price (standard deviation) is 1203 (521) Euro ha⁻¹ a⁻¹.

For deriving reference variable costs for fresh hay transfer, we considered the work steps of cutting, windrowing, collecting, transporting, unloading and distributing fresh hay (Kirmer et al. 2012, Staub et al. 2015). Moreover, we assumed the use of machinery, a distance of 5 km between the farm and donating and reseeded grassland sites as well as between sites, a hay transfer ratio of 1:1 (Staub et al. 2015) and no site preparation. We only considered cost for fuel, labor and opportunity costs, i.e. compensation payment/forgone revenues. The total variable costs of 427 Euro ha⁻¹ a⁻¹ ((0.67 h ha⁻¹ + 0.53 h ha⁻¹ + 5.27 h ha⁻¹ + 3.56 h ha⁻¹ + 0.89 h ha⁻¹) × 13 Euro h⁻¹ + (4.85 l ha⁻¹ + 3.18 l ha⁻¹ + 19.74 l ha⁻¹ + 12.67 l ha⁻¹ + 5.66 l ha⁻¹) × 0.75 Euro l⁻¹ + 250 Euro ha⁻¹) are derived from KTBL (2019) and Kirmer et al. (2012). Note that we always assumed a distance of 5 km between farm and grassland site and that all costs can change with equipment used, distances and other factors.

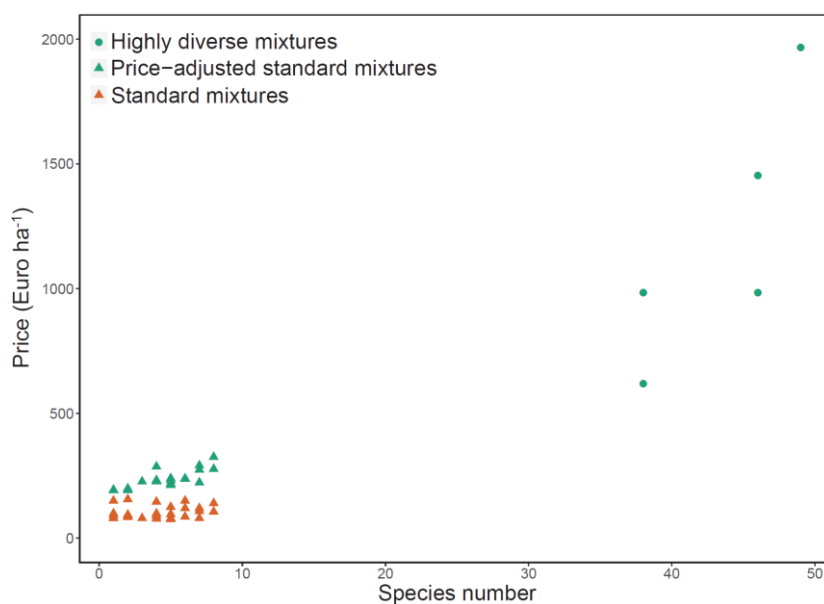


Figure 7: Relationship of seed mixtures prices (Euro ha⁻¹) and number of species in the mixture. The green cycles are highly diverse mixture (38 to 49 species). Orange triangles are standard mixtures (1-8 species). Green triangles are price-adjusted standard mixtures, based on seed costs of the shop focusing on highly diverse regional mixtures.

3.6 Data Availability

The data used in this study is available at Schaub et al. (2019). The source data underlying Fig. 1, Fig. 2 and Fig. 4 to Fig. 7 are provided as a Source Data file. Fig. 3 is based on Table A3 to A8.¹⁷

3.7 Code Availability

The R code (Supplementary Code 1) and Stata code (Supplementary Code 2 and Supplementary Code 3) used in this study are available online.¹⁸

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¹⁷ See Schaub, S., Finger, R., Leiber, F., Probst, S., Kreuzer, M., Weigelt, A., Buchmann, N. & Scherer-Lorenzen, M. (2020). Plant diversity effects on forage quality, yield and revenues of semi-natural grasslands. *Nature Communications*, **11**, 1-11. <https://doi.org/10.1038/s41467-020-14541-4>

¹⁸ See Schaub, S., Finger, R., Leiber, F., Probst, S., Kreuzer, M., Weigelt, A., Buchmann, N. & Scherer-Lorenzen, M. (2020). Plant diversity effects on forage quality, yield and revenues of semi-natural grasslands. *Nature Communications*, **11**, 1-11. <https://doi.org/10.1038/s41467-020-14541-4>

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3.10 Appendix

Table A1: Management intensities. Management intensities were established on subplots within larger experimental plots except the less intensive management that represents the management intensity of the large plots. Cutting frequency is given in cuts per year, all fertilization levels are given in $\text{kg ha}^{-1} \text{a}^{-1}$. Nitrogen was applied as $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ in equal proportions, phosphorus as $\text{P}_2\text{O}_5\text{-P}$ and Potassium as $\text{K}_2\text{O-K}$.

Management intensity	Cutting frequency	Fertilization		
		N	P	K
Extensive	1	0	0	0
Less intensive	2	0	0	0
Intensive	2	100	43.6	83
Highly intensive	4	100	43.6	83
Very highly intensive	4	200	87.2	166

Table A2: Variable description: biomass yield, forage quality and revenues. The respective quality-adjusted yield variables are calculated by multiplying the below mentioned variables with biomass yield.

Name	Description
Biomass yield (g m^{-2})	Biomass is the dry weight of plants of an area at a given time.
Organic matter content (g kg^{-1})	Organic matter is the part of biomass that is not ash.
Neutral detergent fiber content (g kg^{-1})	Neutral detergent fiber describes the total fiber content in forage.
Crude protein content (g kg^{-1})	Crude protein includes true protein and other N-containing compounds and is calculated by convention as crude protein = nitrogen \times 6.25.
(Metabolically) Utilizable crude protein content (g kg^{-1})	Utilizable crude protein reflects the crude protein from microbial synthesis plus the ruminally undegradable protein which is available in the duodenum of ruminants.
Metabolizable energy content (MJ kg^{-1})	Metabolizable energy is the energy available in the metabolism of the animal to gain body weight or to produce milk.
Net energy of lactation (MJ kg^{-1})	Net energy of lactation represents the energy available for lactation (Net energy of lactation = metabolizable energy content – heat from digestion and metabolism).
Milk production potential (kg kg^{-1})	Milk production potential describes the amount of milk which theoretically can produced from the respective forage. It is based on optimal production condition and a cow with 630 kg of live-weight and an annual milk yield of 7000 kg.
Revenues (Euro ha^{-1})	Revenues = Milk production potential yield \times milk price

Source: Ball et al. 2001, Barnes et al. 2003, Edmunds et al. 2012 and Jans et al. 2015.

Mixed Effect Model Results, Model with Legume Presence

Table A3: Result of mixed effect model for annual average metabolizable energy content (MJ kg^{-1}), annual metabolizable energy yield ($\text{MJ m}^{-2} \text{a}^{-1}$), annual average milk production potential ($\text{kg kg}^{-1} \text{a}^{-1}$), annual milk production potential yield ($\text{kg m}^{-2} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Metabolizable energy content	Metabolizable energy yield	Milk production potential	Milk production potential yield
Constant	8.15 (27.45)***	0.32 (1.29)	1.49 (23.93)***	0.06 (1.47)
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)				
Extensive	-0.13 (-1.84)	0.68 (5.95)***	-0.03 (-1.95)	0.12 (5.9)***
Less intensive	-0.09 (-2)	0.81 (5.73)***	-0.02 (-2.01)	0.15 (5.69)***
Intensive	-0.12 (-2.73)*	1 (5.47)***	-0.03 (-2.73)*	0.18 (5.34)***
Highly intensive	0 (0.01)	0.66 (9.87)***	0 (-0.03)	0.13 (9.17)***
Very highly intensive	0.06 (1.42)	0.91 (4.73)***	0.01 (1.4)	0.18 (4.83)***
M				
Less intensive	0.62 (1.99)	1.21 (2.91)*	0.13 (2.04)	0.22 (2.99)*
Intensive	1 (3.58)**	3.08 (11.06)***	0.22 (3.63)**	0.57 (10.64)***
Highly intensive	2.08 (5.79)***	3.41 (24.14)***	0.46 (6.1)***	0.65 (21.31)***
Very highly intensive	2.34 (7.47)***	5.46 (54.88)***	0.52 (7.88)***	1.04 (66.86)***
Legumes (L) \times M				
Extensive	0.48 (2.56)	1.52 (5.66)***	0.11 (2.61)	0.29 (5.86)***
Less intensive	0.76 (4.42)***	4.3 (7.13)***	0.17 (4.47)***	0.82 (7.25)***
Intensive	0.57 (5.43)***	2.25 (4.52)***	0.13 (5.75)***	0.44 (4.65)***
Highly intensive	0.52 (1.68)	2.32 (2.66)*	0.12 (1.66)	0.45 (2.73)*
Very highly intensive	0.16 (0.74)	0.73 (0.95)	0.04 (0.74)	0.14 (0.96)
Functional groups (FG)				
Grasses (G)	-0.1 (-0.82)	-0.87 (-2.36)	-0.03 (-0.91)	-0.17 (-2.49)
Tall Herbs (H)	0.11 (1.44)	1.78 (2.63)	0.03 (1.63)	0.34 (2.74)*
	-0.19 (-0.91)	0.21 (0.43)	-0.04 (-0.89)	0.04 (0.41)
Random effects {Variance components}				
Block (B)	0.004 {0.01}	<0.0001 {<0.0001}	0.0002 {0.0002}	<0.0001 {<0.0001}
Plot (P)	0.08 {0.02}	0.92 {9.04}	0.004 {0.0009}	0.03 {0.04}
Residual	0.43 {0.04}	3.64 {2.7}	0.02 {0.002}	0.13 {0.02}
Number of groups				
Block	4	4	4	4
Plot	79	79	79	79

Table A4: Result of mixed effect model for annual average crude protein content (g kg^{-1}), annual crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$), annual average utilizable crude protein content (g kg^{-1}) and annual utilizable crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Crude protein content	Crude protein yield	Utilizable crude protein content	Utilizable crude protein yield
Constant	121.29 (14.14)***	5.9 (1.59)	182.3 (25.52)***	22.07 (4.6)***
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)				
Extensive	-5.47 (-6.72)***	9.73 (6.72)***	-	-
Less intensive	-4.48 (-1.7)	10.45 (6.22)***	-4.16 (-1.8)	11.66 (10.63)***
Intensive	-2.59 (-6.46)***	14.77 (4.28)***	-2.55 (-1.46)	12.8 (6.14)***
Highly intensive	-0.63 (-0.31)	11.28 (6.7)***	-0.91 (-0.26)	5.87 (2.04)
Very highly intensive	0.78 (0.33)	17.26 (4.64)***	-0.59 (-0.19)	9.84 (3.96)***
M				
Less intensive	16.4 (2.08)	19.28 (4.02)***	-	-
Intensive	18.03 (5.6)***	38.94 (8.37)***	6.83 (0.91)	23.45 (4.25)***
Highly intensive	74.73 (9.11)***	65.9 (15.82)***	50.11 (6.28)***	-0.91 (-0.26)
Very highly intensive	96.24 (7.78)***	109.9 (12.22)***	63.77 (4.62)***	8.75 (1.96)
Legumes (L) \times M				
Extensive	70.04 (18.77)***	35.78 (12.95)***	-	-
Less intensive	81.87 (16.33)***	101.22 (11.55)***	18.9 (2.91)*	79.47 (6.38)***
Intensive	63.64 (5.95)***	64.07 (4.75)***	-1.71 (-0.36)	45.77 (2.81)*
Highly intensive	52.45 (11.17)***	54.23 (5.3)***	2.78 (0.24)	36.16 (4.79)***
Very highly intensive	49.68 (5.07)***	32.99 (1.75)	-5.24 (-0.52)	29.65 (2.5)
Functional groups (FG)				
Grasses (G)	-17.03 (-3.35)**	-19.5 (-3.32)**	12.67 (3.05)*	-15.15 (-4)***
Tall Herbs (H)	-1.75 (-0.27)	25.19 (3.55)**	-39.28 (-5.27)***	18.73 (7.45)***
	5.76 (2.34)	8.31 (0.79)	-14.36 (-3.51)**	1.59 (0.21)
Random effects {Variance components}				
Block (B)	33.3 {35.84}	<0.0001 {<0.0001}	46.18 {40.24}	31.79 {28.21}
Plot (P)	258.94 {88.18}	341.81 {184.54}	114.34 {38.19}	340.74 {363.45}
Residual	480.46 {54.85}	1129.88 {190.35}	474.51 {93.55}	923.77 {127.71}
Number of groups				
Block	4	4	4	4
Plot	80	80	77	77

Note utilizable crude protein content was only measured for the first cut of the year.

Table A5: Result of mixed effect model for annual average organic matter content (g kg^{-1}), annual organic matter yield ($\text{g m}^{-2} \text{a}^{-1}$), annual average neutral detergent fiber content (g kg^{-1}) and annual neutral detergent fiber yield ($\text{g m}^{-2} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Organic matter content	Organic matter yield	Neutral detergent fiber content	Neutral detergent fiber yield
Constant	896.51 (171.91)***	9.76 (0.4)	489.13 (23.16)***	-9.97 (-0.89)
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)				
Extensive	3.14 (5.04)***	85.93 (6.48)***	24.99 (5.29)***	64.71 (6.57)***
Less intensive	2.86 (2.74)*	92.22 (6.04)***	13.85 (2.82)*	65.54 (5.72)***
Intensive	2.5 (2.23)	117.66 (6.31)***	18.65 (5.2)***	85.12 (10.15)***
Highly intensive	1.66 (0.83)	66.78 (21.1)***	3.24 (0.92)	28.74 (10.53)***
Very highly intensive	0.71 (1.43)	73.08 (4.08)***	8.34 (2.34)	40.88 (4.06)***
M				
Less intensive	-0.43 (-0.07)	131.32 (2.75)*	-24.46 (-3.7)**	71.9 (2.1)
Intensive	0.89 (0.14)	308.88 (10.77)***	-28.58 (-1.5)	176.09 (5.81)***
Highly intensive	-17.58 (-2.61)	277.17 (12.56)***	-136.34 (-6.88)***	138.31 (10.93)***
Very highly intensive	-10.97 (-4.65)***	479.81 (23.84)***	-126.47 (-5.87)***	226.28 (17.93)***
Legumes (L) $\times M$				
Extensive	35.68 (8.5)***	143.25 (5.44)***	52.99 (2.01)	122.45 (4.92)***
Less intensive	34.59 (15.31)***	402.99 (7.58)***	68.15 (6.91)***	251.12 (8.31)***
Intensive	28.22 (8.03)***	169.82 (3.1)*	62.41 (3.1)*	107.41 (2.98)*
Highly intensive	32.05 (10.41)***	153.32 (2.85)*	71.68 (3.73)**	115.56 (3.07)*
Very highly intensive	26.87 (13.59)***	54.7 (0.78)	54.94 (3.33)**	42.09 (0.98)
Functional groups (FG)				
Grasses (G)	-16.5 (-6.9)***	-74.73 (-1.94)	-68.82 (-5.46)***	-74.86 (-2.69)*
Tall Herbs (H)	34.08 (8.93)***	167.63 (2.48)	189.36 (10.33)***	170.94 (3.23)**
Random effects {Variance components}				
Block (B)	18.42 {15.43}	<0.0001 {<0.0001} 12675.99	315.95 {261.11}	106.47 {465.46}
Plot (P)	207.4 {53.25}	{4902.78}	1508.65 {131.07}	5140.04 {1952.37}
Residual	195.26 {26.3}	34959.5 {2679.35}	3394.83 {469.45}	13461.4 {1771.81}
Number of groups				
Block	4	4	4	4
Plot	79	79	79	79

Table A6: Result of mixed effect model for annual biomass yield ($\text{g m}^{-2} \text{a}^{-1}$) and annual revenues (Euro $\text{ha}^{-1} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Biomass yield	Revenues
Constant	10.82 (0.41)	200.7 (1.47)
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)		
Extensive	92.66 (6.5)***	378.36 (5.9)***
Less intensive	100.09 (6.1)***	453.5 (5.69)***
Intensive	128.28 (6.18)***	562.46 (5.34)***
Highly intensive	74.25 (23.3)***	388.47 (9.17)***
Very highly intensive	81.67 (4.08)***	547.66 (4.83)***
M		
Less intensive	148.26 (2.95)**	691.78 (2.99)*
Intensive	343.25 (11.04)***	1776.15 (10.64)***
Highly intensive	318.68 (13.57)***	2011.52 (21.31)***
Very highly intensive	543.28 (25.27)***	3231.98 (66.86)***
Legumes (L) \times M		
Extensive	148.33 (5.43)***	893.45 (5.86)***
Less intensive	425.88 (7.33)***	2535.63 (7.25)***
Intensive	174.73 (3.14)**	1356.02 (4.65)***
Highly intensive	155.77 (2.69)**	1396.89 (2.73)*
Very highly intensive	50.6 (0.69)	439.61 (0.96)
Functional groups (FG)	-74.59 (-1.72)	-523.67 (-2.49)
Grasses (G)	167.57 (2.22)*	1064.35 (2.74)*
Tall Herbs (H)	24.82 (0.4)	119.7 (0.41)
Random effects {Variance components}		
Block (B)	<0.0001 {<0.0001}	<0.0001 {<0.0001}
Plot (P)	14603.1 {5673.38}	296129.29 {160445.45}
Residual	41343.77 {3176.59}	1238636.63 {137002.29}
Number of groups		
Block	4	4
Plot	80	79

Differences between Plant Diversity Effects, Model with Legume Presence

Table A7: Differences and the corresponding significance levels of the differences between plant diversity effects per management intensity on annual metabolizable energy yield ($\text{MJ m}^{-2} \text{a}^{-1}$), annual milk production potential yield ($\text{kg m}^{-2} \text{a}^{-1}$) and annual biomass yield ($\text{g m}^{-2} \text{a}^{-1}$). 'Management intensity - management intensity' indicates the compared management intensities. *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Numbers in parentheses are Wald χ^2 statistics.

Management intensity - management intensity	Metabolizable energy yield	Milk production potential yield	Biomass yield
Extensive - less intensive	-0.13 (0.39)	-0.02 (0.43)	-7.43 (0.1)
Extensive intensive	-0.32 (3.13)	-0.06 (3.32)	-35.63 (2.1)
Extensive - highly intensive	0.02 (0.03)	-0.003 (0.01)	18.41 (1.42)
Extensive - very highly intensive	-0.23 (2.55)	-0.05 (3.89)	10.99 (0.71)
Less intensive - intensive	-0.19 (2.49)	-0.04 (2.3)	-28.19 (5.86)*
Less intensive - highly intensive	0.15 (0.58)	0.02 (0.31)	25.84 (2.82)
Less intensive - very highly intensive	-0.11 (0.32)	-0.03 (0.71)	18.42 (0.93)
Intensive - highly intensive	0.34 (1.98)	0.06 (1.45)	54.03 (8.56)**
Intensive - very highly intensive	0.09 (0.22)	0.005 (0.02)	46.61 (4.75)*
Highly intensive - very highly intensive	-0.26 (1.06)	-0.05 (1.13)	-7.42 (0.13)

Table A8: Differences and the corresponding significance levels of the differences between plant diversity effects per management intensity on annual crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$), annual organic matter yield ($\text{g m}^{-2} \text{a}^{-1}$), annual neutral detergent fiber yield ($\text{g m}^{-2} \text{a}^{-1}$) and annual revenues ($\text{Euro ha}^{-1} \text{a}^{-1}$). 'Management intensity - management intensity' indicates the compared management intensities. *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Numbers in parentheses are Wald χ^2 statistics.

Management intensity - management intensity	Crude protein yield	Organic matter yield	Neutral detergent fiber yield	Revenues
Extensive - less intensive	-0.71 (0.07)	-6.28 (0.08)	-0.83 (0.004)	-75.14 (0.43)
Extensive intensive	-5.04 (3.12)	-31.73 (1.95)	-20.42 (3.27)	-184.1 (3.32)
Extensive - highly intensive			35.97	
	-1.55 (0.3)	19.15 (1.57)	(10.72)**	-10.11 (0.01)
Extensive - very highly intensive			23.83	
	-7.53 (5.43)	12.85 (1.28)	(31.05)***	-169.31 (3.89)
Less intensive - intensive	-4.32 (1.8)	-25.45 (5.37)	-19.59 (8.45)*	-108.95 (2.3)
Less intensive - highly intensive	-0.84 (0.21)	25.43 (3.28)	36.8 (9.58)*	65.03 (0.31)
Less intensive - very highly intensive	-6.81 (2.5)	19.13 (1.21)	24.66 (4.93)	-94.16 (0.71)
Intensive - highly intensive			56.38	
	3.49 (0.88)	50.88 (9.49)*	(57.3)***	173.99 (1.45)
Intensive - very highly intensive			44.25	
	-2.49 (0.23)	44.58 (5.19)	(16.1)***	14.79 (0.02)
Highly intensive - very highly intensive	-5.98 (1.24)	-6.3 (0.11)	-12.14 (1.06)	-159.19 (1.13)

Robustness Analysis – Mixed Effect Model Results, Model with Legume Share

Table A9: Robustness analysis (model includes legume shares) – Result of mixed effect model for annual average metabolizable energy content (MJ kg^{-1}), annual metabolizable energy yield ($\text{MJ m}^{-2} \text{a}^{-1}$), annual average milk production potential ($\text{kg kg}^{-1} \text{a}^{-1}$), annual milk production potential yield ($\text{kg m}^{-2} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Metabolizable energy content	Metabolizable energy yield	Milk production potential	Milk production potential yield
Constant	8.03 (25.78)***	0.06 (0.28)	1.47 (22.58)***	0.01 (0.39)
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)				
Extensive	-0.13 (-1.65)	0.64 (4.58)***	-0.03 (-1.74)	0.11 (4.51)***
Less intensive	-0.07 (-1.54)	0.97 (6.24)***	-0.02 (-1.55)	0.18 (6.22)***
Intensive	-0.11 (-2.65)*	1.01 (5.55)***	-0.03 (-2.63)	0.18 (5.39)***
Highly intensive	0 (0.1)	0.68 (13.79)***	0 (0.05)	0.13 (12.62)***
Very highly intensive	0.05 (1.24)	0.86 (5.01)***	0.01 (1.23)	0.17 (5.13)***
M				
Less intensive	0.59 (1.98)	0.82 (2.17)	0.13 (2.05)	0.15 (2.23)
Intensive	1 (3.67)**	2.86 (19.33)***	0.22 (3.74)**	0.53 (19.54)***
Highly intensive	2.09 (5.51)***	3.22 (32.3)***	0.46 (5.82)***	0.61 (30.65)***
Very highly intensive	2.39 (7.43)***	5.4 (26.45)***	0.53 (7.85)***	1.03 (28.66)***
Legume share ($LS^{0.5}$) $\times M$				
Extensive	0.61 (2.08)	1.01 (2.05)	0.14 (2.15)	0.19 (2.13)
Less intensive	0.95 (3.35)**	5 (15.18)***	0.22 (3.43)**	0.95 (15.17)***
Intensive	0.7 (3.05)*	2.47 (2.84)*	0.16 (3.18)**	0.48 (2.98)*
Highly intensive	0.65 (1.38)	2.36 (1.8)	0.15 (1.37)	0.47 (1.88)
Very highly intensive	0.02 (0.08)	-0.06 (-0.06)	0.01 (0.09)	-0.01 (-0.05)
Functional groups (FG)				
Grasses (G)	-0.03 (-0.25)	-0.39 (-1.15)	-0.01 (-0.33)	-0.08 (-1.24)
Tall Herbs (H)	0.09 (1.55)	1.46 (2.25)	0.03 (1.77)	0.28 (2.36)
	-0.18 (-0.86)	0.11 (0.19)	-0.04 (-0.84)	0.02 (0.18)
Random effects {Variance components}				
Block (B)	0.004 {0.004}	<0.0001 {<0.0001}	0.0002 {0.0002}	<0.0001 {<0.0001}
Plot (P)	0.08 {0.02}	1.01 {0.42}	0.004 {0.001}	0.03 {0.03}
Residual	0.43 {0.04}	3.63 {0.38}	0.02 {0.002}	0.13 {0.01}
Number of groups				
Block	4	4	4	4
Plot	79	79	79	79

Table A10: Robustness analysis (model includes legume shares) – Result of mixed effect model for annual average crude protein content (g kg^{-1}), annual crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$), annual average utilizable crude protein content (g kg^{-1}) and annual utilizable crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Crude protein content	Crude protein yield	Utilizable crude protein content	Utilizable crude protein yield
Constant	105.84 (13.79)***	-1.51 (-0.64)	178.31 (24.46)***	7.65 (5.13)***
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)				
Extensive	-4.37 (-5.56)***	8.77 (4.28)***	-	-
Less intensive	-2.49 (-1.16)	14.02 (11.03)***	-3.19 (-1.35)	14.25 (18.8)***
Intensive	-1.92 (-4.02)***	15.47 (4.27)***	-3.1 (-2.31)	12.89 (5.46)***
Highly intensive	-0.65 (-0.37)	11.55 (6.97)***	-0.69 (-0.24)	5.89 (2.58)
Very highly intensive	0.93 (0.42)	16.51 (5.27)***	-0.97 (-0.29)	9.38 (4.26)***
M				
Less intensive	14.48 (1.98)	9.61 (2.1)	-	-
Intensive	18.29 (3.52)**	32.29 (6.32)***	8.53 (1.17)	25.85 (4)***
Highly intensive	75.69 (9.55)***	61.44 (15.28)***	53.85 (7.78)***	4.73 (2.07)
Very highly intensive	98.31 (8.72)***	107.37 (11.57)***	67.12 (4.85)***	13.64 (3.07)*
Legume share ($LS^{0.5}$) $\times M$				
Extensive	86.93 (10.06)***	27.13 (3.16)**	-	-
Less intensive	103.87 (10.42)***	123.16 (18.63)***	28.4 (4.9)***	97.42 (8.46)***
Intensive	79.5 (4.11)***	79.09 (3.36)**	2.41 (0.28)	56.86 (2.3)
Highly intensive	64.34 (8.63)***	59.08 (4.04)***	-3.22 (-0.2)	30.28 (2.94)*
Very highly intensive	54.17 (3.37)**	26.15 (1.06)	-10.16 (-0.8)	26.07 (1.81)
Functional groups (FG)				
Grasses (G)	-8.29 (-1.86)	-8.87 (-1.46)	13.01 (3.41)**	-6.98 (-1.93)
Tall Herbs (H)	-2.32 (-0.39)	19.81 (2.49)	-38.52 (-5.85)***	16.15 (4.3)***
Random effects				
{Variance components}				
Block (B)	14.51 {24.17}	<0.0001 {<0.0001}	42.18 {37.41}	19.25 {35.9}
Plot (P)	215.34 {78.69}	361.51 {177.62}	113.92 {38.49}	335.25 {348.56}
Residual	478.08 {52.34}	1108.12 {221.92}	469.55 {91.15}	923.03 {137.42}
Number of groups				
Block	4	4	4	4
Plot	80	80	77	77

Note utilizable crude protein content was only measured for the first cut of the year.

Table A11: Robustness analysis (model includes legume shares) – Result of mixed effect model for annual average organic matter content (g kg^{-1}), annual organic matter yield ($\text{g m}^{-2} \text{a}^{-1}$), annual average neutral detergent fiber content (g kg^{-1}) and annual neutral detergent fiber yield ($\text{g m}^{-2} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Organic matter content	Organic matter yield	Neutral detergent fiber content	Neutral detergent fiber yield
Constant	888.03 (150.09)***	-12.04 (-0.55)	477.61 (20.23)***	-27.32 (-2.94)*
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)				
Extensive	3.66 (5.87)***	83.57 (5.13)***	24.86 (5.38)***	64.71 (5.41)***
Less intensive	3.4 (3.84)***	108.04 (6.86)***	14.9 (2.95)*	74.62 (5.95)***
Intensive	2.61 (2.71)*	116.34 (6.82)***	19.39 (5.03)***	83.66 (12.13)***
Highly intensive	2.06 (1.06)	66.09 (11.99)***	4.49 (1.09)	28.81 (7.32)***
Very highly intensive	0.81 (2.66)*	67.51 (4.27)***	8.23 (2.33)	37.18 (4.31)***
M				
Less intensive	1.14 (0.17)	94.28 (2.05)	-24.48 (-3.63)**	53.15 (1.7)
Intensive	3.1 (0.46)	291.33 (11.22)***	-27.14 (-1.24)	168.96 (5.46)***
Highly intensive	-15.53 (-2.07)	269.41 (10.51)***	-135.98 (-7.38)***	133.22 (8.87)***
Very highly intensive	-8.88 (-3.11)*	473.66 (16.52)***	-127.16 (-6.16)***	223.98 (14.32)***
Legume share ($LS^{0.5}$) $\times M$				
Extensive	46.58 (9.45)***	95.28 (1.8)	55.71 (1.9)	96.89 (2.64)*
Less intensive	40.25 (11.11)***	471.69 (15.43)***	70.75 (10.17)***	283.94 (18.11)***
Intensive	31.47 (7.22)***	178.48 (1.81)	60.28 (2.34)	102.98 (1.59)
Highly intensive	35.62 (5.42)***	115.13 (1.94)	73.45 (3.37)**	95.97 (2.07)
Very highly intensive	29.37 (6.38)***	-18.23 (-0.2)	61.1 (3.31)**	-5.55 (-0.09)
Functional groups (FG)				
Grasses (G)	-11.52 (-4.12)***	-31.06 (-0.86)	-56.5 (-5.2)***	-44.19 (-1.74)
Tall Herbs (H)	32.51 (8.59)***	136.17 (2.18)	181.17 (12.15)***	146.64 (3.07)*
Random effects {Variance components}				
Block (B)	18.13 {14.88}	<0.0001 {<0.0001}	324.17 {267.41}	196.39 {450.2}
Plot (P)	201.53 {59.03}	13428.88 {5914.07}	1565.97 {218.75}	5445.13 {1727.77}
Residual	193.42 {27.65}	34918.21 {3724.26}	3404.91 {460.79}	13541.5 {1983.81}
Number of groups				
Block	4	4	4	4
Plot	79	79	79	79

Table A12: Robustness analysis (model includes legume shares) – Result of mixed effect model for annual biomass yield ($\text{g m}^{-2} \text{a}^{-1}$) and annual revenues ($\text{Euro ha}^{-1} \text{a}^{-1}$). *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Standard errors are corrected for heteroscedasticity. Numbers in parentheses are z-values, which are based on robust standard errors.

	Biomass yield	Revenues
Constant	-11.56 (-0.48)	44.37 (0.39)
Plant diversity ^{0.5} ($D^{0.5}$) \times Management (M)		
Extensive	90.27 (5.24)***	355.41 (4.51)***
Less intensive	117.07 (6.8)***	546.15 (6.22)***
Intensive	126.82 (6.64)***	567.44 (5.39)***
Highly intensive	73.41 (12.59)***	400.23 (12.62)***
Very highly intensive	75.64 (4.27)***	514.24 (5.13)***
M		
Less intensive	109.49 (2.29)*	464.1 (2.23)
Intensive	325.27 (12.65)***	1646.41 (19.54)***
Highly intensive	311.24 (11.52)***	1895.4 (30.65)***
Very highly intensive	537.36 (17.76)***	3195.64 (28.66)***
Legume share ($LS^{0.5}$) \times M		
Extensive	94.88 (1.7)	600.86 (2.13)
Less intensive	495.88 (16.77)***	2955.23 (15.17)***
Intensive	180.23 (1.77)	1496.89 (2.98)*
Highly intensive	110.42 (1.76)	1441.59 (1.88)
Very highly intensive	-31.51 (-0.35)	-27.85 (-0.05)
Functional groups (FG)	-28.03 (-0.69)	-235.3 (-1.24)
Grasses (G)	132.59 (1.9)	871.45 (2.36)
Tall Herbs (H)	12.44 (0.18)	59.64 (0.18)
Random effects {Variance components}		
Block (B)	<0.0001 {<0.0001}	<0.0001 {0.001}
Plot (P)	15488.5 {13392.26}	328136.2 {135751.07}
Residual	41314.4 {4062.21}	1233574.1 {143544.79}
Number of groups		
Block	4	4
Plot	80	79

Differences between Plant Diversity Effects, Model with Legume Presence

Table A13: Robustness analysis (model includes legume shares) – Differences and the corresponding significance levels of the differences between plant diversity effects per management intensity on annual metabolizable energy yield ($\text{MJ m}^{-2} \text{a}^{-1}$), annual milk production potential yield ($\text{kg m}^{-2} \text{a}^{-1}$) and annual biomass yield ($\text{g m}^{-2} \text{a}^{-1}$). ‘Management intensity - management intensity’ indicates the compared management intensities. *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Numbers in parentheses are Wald χ^2 statistics.

Management intensity - management intensity	Metabolizable energy yield	Milk production potential yield	Biomass yield
Extensive - less intensive	-0.32 (2.79)	-0.06 (3.15)	-26.8 (1.24)
Extensive intensive	-0.36 (3.7)	-0.07 (4.02)	-36.55 (2.07)
Extensive - highly intensive	-0.03 (0.03)	-0.014 (0.2)	16.86 (0.83)
Extensive - very highly intensive	-0.21 (4.1)	-0.05 (6.86)	14.62 (1.74)
Less intensive - intensive	-0.04 (0.61)	-0.01 (0.45)	-9.75 (6.12)*
Less intensive - highly intensive	0.29 (2.27)	0.05 (1.66)	43.66 (11.29)***
Less intensive - very highly intensive	0.11 (0.46)	0.01 (0.12)	41.42 (6.28)*
Intensive - highly intensive	0.33 (2.29)	0.05 (1.65)	53.41 (14.44)***
Intensive - very highly intensive	0.15 (0.81)	0.017 (0.31)	51.17 (7.53)**
Highly intensive - very highly intensive	-0.18 (0.69)	-0.04 (0.76)	-2.24 (0.02)

Table A14: Differences and the corresponding significance levels of the differences between plant diversity effects per management intensity on annual crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$), annual organic matter yield ($\text{g m}^{-2} \text{a}^{-1}$), annual neutral detergent fiber yield ($\text{g m}^{-2} \text{a}^{-1}$) and annual revenues ($\text{Euro ha}^{-1} \text{a}^{-1}$). 'Management intensity - management intensity' indicates the compared management intensities. *, **, *** denote significance at the 5%, 1% and 0.1% level, respectively (corrected for multiple comparisons). Numbers in parentheses are Wald χ^2 statistics.

Management intensity - management intensity	Crude protein yield	Organic matter yield	Neutral detergent fiber yield	Revenues
Extensive - less intensive	-5.25 (6.1)	-24.47 (1.19)	-9.92 (0.486)	-190.75 (3.15)
Extensive intensive	-6.7 (4.13)	-32.77 (1.93)	-18.95 (2.26)	-212.04 (4.02)
Extensive - highly intensive	-2.78 (0.71)	17.48 (0.91)	35.9 (8.3)*	-44.83 (0.2)
Extensive - very highly intensive	-7.74 (12.36)**	16.06 (2.68)	27.53 (31.03)***	-158.83 (6.86)
Less intensive - intensive	-1.45 (0.33)	-8.3 (4.6)	-9.04 (1.87)	-21.29 (0.45)
Less intensive - highly intensive	2.47 (2.71)	41.95 (12.94)**	45.82 (18.95)***	145.92 (1.66)
Less intensive - very highly intensive	-2.49 (0.51)	40.53 (7.24)*	37.45 (11.97)**	31.91 (0.12)
Intensive - highly intensive	3.92 (1.5)	50.25 (16.96)***	54.85 (187.53)***	167.21 (1.65)
Intensive - very highly intensive	-1.04 (0.04)	48.83 (8.17)*	46.48 (25.27)***	53.2 (0.31)
Highly intensive - very highly intensive	-4.97 (1.12)	-1.42 (0.01)	-8.37 (0.77)	-114.01 (0.76)

Table A15: Pearson correlation coefficient of annual biomass yield ($\text{g m}^{-2} \text{a}^{-1}$) and annual metabolizable energy yield ($\text{MJ m}^{-2} \text{a}^{-1}$), annual milk production potential yield ($\text{kg m}^{-2} \text{a}^{-1}$), annual crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$), annual utilizable crude protein yield ($\text{g m}^{-2} \text{a}^{-1}$), annual organic matter yield ($\text{g m}^{-2} \text{a}^{-1}$) and annual neutral detergent fiber yield ($\text{g m}^{-2} \text{a}^{-1}$). Note utilizable crude protein content was only measured for the first cut of the year, therefore, the Pearson correlation coefficient refers to biomass yield and utilizable crude protein yield of the first cut.

	Biomass yield
Metabolizable energy yield	0.98
Milk production potential yield	0.97
Crude protein yield	0.88
Utilizable crude protein yield	0.96
Organic matter yield	1.00

Table A16: Cutting dates in 2007. Biomass yield samples of dates given in italics were only quantified and not analyzed for forage quality variables. See Table A2 for details on management intensities.

Management intensity	Cutting season			
	Spring		Autumn	
Extensive	--	--	--	September 1
Less intensive	June 4	--	--	September 1
Intensive	June 4	--	--	September 1
Highly intensive	April 24	<i>June 4</i>	<i>July 23</i>	September 1
Very highly intensive	April 24	<i>June 4</i>	<i>July 23</i>	September 1

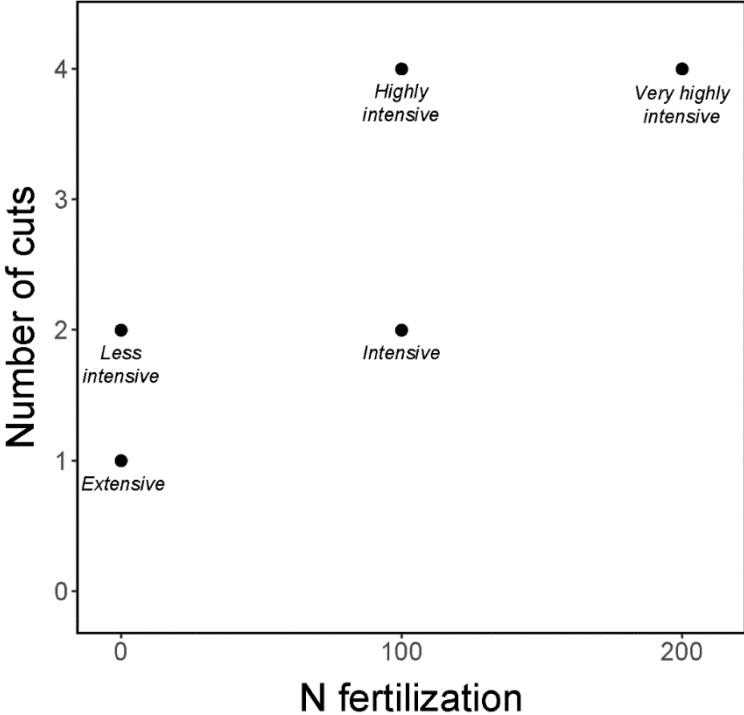


Figure A1: Illustration of the management intensities and their elements (cutting frequency and N fertilization (kg ha⁻¹ a⁻¹) per year) tested in our study. The experimental design includes five different management intensities, which are here defined as: extensive, less intensive, intensive, highly intensive and very highly intensive management. Beside N, the fertilization included P and K (see Table A2).

Appendix References

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4 Chapter 4 – Economics of Species Diversity in Grasslands under Increasing Drought Risks¹⁹

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Abstract

Abstract: Droughts adversely affect grassland production. Increasing drought risks due to climate change are predicted to cause farm income losses and income risk increases. We investigate grassland species diversity as risk management instrument to mitigate adverse drought effects on hay yields. In our theoretical and empirical (econometric and simulation) analysis we investigate the optimal species diversity choice under increasing drought risks. In the analysis, we consider risk aversion, differences in farm types and market responses to droughts. Our results show that increasing drought risks negatively affected most farmers, with the effect size depending on farm type. Species diversity increased farm income and decreased income risk independent of the drought probability. Our theoretical and empirical results consistent in most cases. However, they differ in some cases, as positive drought effects on hay prices compensated drought induced hay yield losses and as species diversity effects did not differ between drought and non-drought condition in our sample. We conclude that species diversity should be taken into consideration in the sustainable management of grasslands as it increases expected income and reduces risk – also under predicted climate change condition.

Keywords: drought, weather risk, species diversity, biodiversity, hay price, yield, sustainable intensification

¹⁹ An earlier version of this chapter was presented at the 19th BIOECON (Biodiversity and Economics for Conservation) Conference.

4.1 Introduction

In the agricultural sector, farmers' decisions are taken in the context of high uncertainty and risk, strongly driven by weather variability and extreme weather events (Porter and Semenov 2005, Schlenker and Roberts 2009). In particular, droughts are critical for agricultural production, which are expected to increase in frequency and magnitude under climate change (Dai 2011, Dai 2013, IPCC 2013, Spinoni et al. 2018, Webber et al. 2018, Grillakis 2019). This leads to 'increasing drought risks', which can strongly affect grasslands that are important forage sources for animal production (Huyghe et al. 2014, Barnes et al. 2003, Finger et al. 2013, Ma et al. 2017). For example, the 2018 summer drought in Europe considerably reduced hay yields, increased hay prices and even caused farmers to slaughter or sell livestock earlier than in other years (European Parliament 2018, Lucy Hooker 2018). Here, we focus on the consequences of increasing drought risks for farmers and the role of species diversity in grasslands as strategy to reduce farmers' risk exposure against increasing drought risks. Species diversity is hypothesized to reduce adverse effects of perturbations, such as droughts, on yields and their stability (Yachi and Loreau 1999). In the context of species diversity, we investigate if and how farmers can use species diversity choice as strategic risk instrument in grassland management. Previous research showed that species diversity can reduce variability of hay yields (see e.g. Hooper et al. 2005, Isbell et al. 2009), even when accounting for hay quality (Schaub et al. 2020a). Additionally, empirical studies explicitly addressing droughts in grasslands showed that species diversity can reduce adverse drought effects on hay yields (Van Ruijven and Berendse 2010, Vogel et al. 2012, Isbell et al. 2015, Hofer et al. 2016, Haughey et al. 2018). However, economic assessments of the effects of species diversity (henceforth 'diversity effects') as risk management strategy have so far been restricted to the field-level and did not explicitly account for drought risks (Schläpfer et al. 2002, Koellner and Schmitz 2006, Finger and Buchmann 2015, Binder et al. 2018, Schaub et al. 2020a). Nonetheless, to assess adverse effects of increasing drought risks and the use of risk management instruments such as species diversity, the integration of farm-level context and the consideration of (hay) market responses to droughts are needed.

We add to the existing literature in several ways. First, we provide a theoretical model to investigate the impact of increasing drought risks and the optimal species diversity choice in response to increasing drought risks as well as the implication of farmers' risk preferences in this setting. We include a farm-level perspective, i.e. hay use for milk production, and we account for hay market responses to drought shocks. A particular focus of our paper is on the possible implications of increasing drought risks and species diversity on farm management for different farm types. More specifically, we distinguish how farmers use their hay, i.e. if farmers are net sellers or net buyers of hay. Second, we provide empirical evidence of drought and diversity effects on hay yields and their variability as well as of drought effects on hay price distributions. To this end, we use a unique combination of field experiments and hay price

data. Finally, we integrate the theoretical and empirical components to simulate implications of different levels of increasing drought risks and species diversity choices considering various levels of risk aversion.

Our results show that increasing drought risks negatively affected most farmers, with the effect size depending on farm type. Moreover, we find that species diversity can be an important management instrument to increase expected income while decreasing income risk.

The remainder of this article is organized as follows: First, we develop an economic model considering interactions of increasing drought risks, hay yields, hay prices and species diversity. Second, we perform a theoretical analysis of the processes described in our economic model. Third, we develop an empirical framework. This is followed by a description of the hay yield, hay price and weather data as well as the empirical results. Furthermore, we conduct a simulation using our empirical findings. Finally, we discuss and conclude on our findings.

4.2 Economic Model

We develop in this section of the paper a stylized economic model for investigating the adverse effects of increasing drought risks for farmers as well as for using species diversity as a risk management instrument in grasslands to mitigate adverse drought effects. To this end, we focus on drought and diversity effects on hay yields and prices derived from previous theoretical and empirical studies.

Our model comprises farmers that are either net sellers or net buyers of hay. For illustration purposes, we focus on two distinct types of farmers: either farmers that manage grasslands to only sell hay (= net sellers) or farmers that manage grasslands and always purchase additional hay to feed their ruminants for milk production (= net buyers).²⁰ The farmers generate revenue by selling their produce (hay and/or milk) at its market price, p_j . For milk producing net sellers only excess hay is sold and for milk producing net buyers additional is bought.²¹ Thus,

$$R_{\theta} = (y_{hay} - d_{hay})p_{hay} + y_{milk} p_{milk} \quad (1)$$

where R_{θ} represents revenues of either net sellers or net buyers depending on θ . y_{hay} and y_{milk} are produced hay and milk, respectively, d_{hay} is farmers' hay demand and p_{hay} and p_{milk} are hay and

²⁰ Note that in reality, varying degrees exist of net buyers and net sellers of hay. We here focus on two types of farms to ensure clarity.

²¹ We assume that farmers only balance their hay deficits by buying hay, reflecting the farm focus to minimize or abandon the use of concentrate feed. In Switzerland, for example, subsidies are linked to maximum use of concentrate of total feed (Mack and Huber 2017). In other countries like Austria, special 'hay milk' labeled products are on the market (see e.g. Paredes et al. 2018).

milk market prices, respectively. Additionally, hay and milk production cause costs, C_θ , which are deterministic, e.g. for seeds and fertilizer. The income of the two farm types is defined as follows:

$$\pi_\theta = R_\theta - C_\theta \quad (2)$$

$E(y_j)$, $E(p_j)$, $E(R_\theta)$ and $E(\pi_\theta)$ are the respective expected values. $Var(y_j)$, $Var(p_j)$, $Var(R_\theta)$ and $Var(\pi_\theta)$ are the respective variances. Note that while we assume that hay yields and hay prices are stochastic, we assume that the conversion of feed into milk, the intra-annual demand for hay as well as milk prices are deterministic. In the analysis we focus on the farm-level and do not consider independences between farms.

4.2.1 Effects of Droughts and Species Diversity

Stochastic environmental shocks such as droughts or pest infestation contribute to the observed variability of hay yields. Droughts are defined as a situation when plant available water is below a critical threshold over a specific period. Water availability depends on precipitation and evapotranspiration (Vicente-Serrano et al. 2010). In our analysis, we focus on effects of increasing drought probabilities, i.e. increasing drought risks. The drought probability is defined as k henceforward. The drought risk is defined by the probability that an index for water availability (see e.g. Tian et al. 2018) falls below a specific threshold (see e.g. Trenberth et al. 2014, Yu et al. 2014, Isbell et al. 2015). In this study, we use for identifying droughts in the hay price time period the Standardized Precipitation Evapotranspiration Index (SPEI) as an index of water availability and a threshold that indicates natural ‘severe droughts’ (Yu et al. 2014). Under increasing drought risks, the drought probability, e.g. that the SPEI is below a certain threshold, increases compared to the baseline scenario. We assume that increasing drought risks reduce expected hay yields and increases their variance (Barnes et al. 2003, Ma et al. 2017),²² i.e.:

$$\frac{\partial E(y_{hay}(k))}{\partial k} < 0 \quad (3)$$

$$\frac{\partial Var(y_{hay}(k))}{\partial k} > 0 \quad (4)$$

Farmers have a set of management instruments to influence hay production such as different farm inputs (e.g. irrigation, nitrogen) as well as the level of species diversity, v . Higher species diversity can be achieved by farmers for instance, with sowing or over-sowing of grasslands with mixtures as well as by adjusting fertilization and cutting regimes (see e.g. Walker et al. 2004). Species diversity is

²² Our analysis focuses on the role of species diversity to cope with increasing drought risks. The first part of the paper focusses on the theoretical framework, while the second part will focus on the empirical relationships.

assumed to generally increase hay yields and it reduces variability of hay yields, while both effects reveal saturation (see e.g. Tilman et al. 1996, Hooper et al. 2005, Isbell et al. 2009, Marquard et al. 2009, Finn et al. 2013, Finger and Buchmann 2015, Schaub et al. 2020b). Hence:

$$\frac{\partial E(y_{hay}(k,v))}{\partial v} > 0 \ \& \ \frac{\partial^2 E(y_{hay}(k,v))}{\partial v^2} \leq 0 \quad (5)$$

$$\frac{\partial Var(y_{hay}(k,v))}{\partial v} < 0 \ \& \ \frac{\partial^2 Var(y_{hay}(k,v))}{\partial v^2} \geq 0 \quad (6)$$

Moreover, in the event of droughts, species diversity is hypothesized to reduce hay yield losses and to decrease variability of hay yields, yet with diminishing returns (Yachi and Loreau 1999), i.e.:

$$\frac{\partial^2 E(y_{hay}(k,v))}{\partial k \partial v} > 0 \ \& \ \frac{\partial^3 E(y_{hay}(k,v))}{\partial k \partial v^2} \leq 0 \quad (7)$$

$$\frac{\partial Var(y_{hay}(k,v))}{\partial k \partial v} < 0 \ \& \ \frac{\partial^3 Var(y_{hay}(k,v))}{\partial k \partial v^2} \geq 0 \quad (8)$$

Droughts are systemic and thus decrease the aggregated hay supply in a region, Y , which is the sum of hay yields of individual farmers in a region ($Y_{hay} = \sum_{i=1}^n y_{hay,i}$). Given the negative relationship of aggregated hay supply and hay market prices, droughts leads to an increase in hay prices (Schaub and Finger 2020). Moreover, as droughts increase variance of hay yields, they also increase variance of aggregated hay supply, thus, the variance of hay prices. Therefore:

$$\frac{\partial E(p_{hay}(k,Y))}{\partial k} > 0 \quad (9)$$

$$\frac{\partial Var(p_{hay}(k,Y))}{\partial k} > 0 \quad (10)$$

Within a region, the hay yields of a farm and the regional aggregated supply are assumed to be positively but not perfectly correlated (McKinnon 1967). Hence, the hay yields of a farmer are negatively correlated with hay prices; this mechanism is called ‘natural hedge’ and it mitigates the income risk. The natural hedge is often found to be ‘imperfect’, i.e. $-1 < cor(y_{hay}, p_{hay}) < 0$ (see e.g. Finger 2012, Ramsey et al. 2019). We assume that farmers are price takers and their decisions do not affect the regional aggregated hay supply. We thus neglect possible effects of species diversity choices on hay prices.

Costs of hay production in our models are a function of species diversity, with the properties:

$$\frac{\partial c_{hay}(v)}{\partial v} > 0 \ \& \ \frac{\partial^2 c_{hay}(v)}{\partial v^2} \geq 0 \quad (11)$$

Farmers decide on the species diversity level in the beginning of the year so that these costs are deterministic. By deciding on a specific species diversity level farmers select a specific ‘lottery’ (Crocker and Shogren 2001, Baumgärtner 2007), i.e. a lottery with a specific mean and variance of income.

4.2.2 Farmers Optimal Choice of Species Diversity

Farmers are assumed to maximize their expected utility and to be risk averse, which is often observed (see e.g. Iyer et al. 2020). Thus, we consider farmers’ von Neumann-Morgenstern utility function, $U(\cdot)$, to be monotonic ($U' > 0$) and to reflect risk aversion ($U'' < 0$).

In the expected utility framework, the cost of risk bearing by an individual can be determined by the risk premium, RP (see e.g. Chavas 2004). The risk premium represents the amount that leaves a farmer indifferent between stochastic income from hay production and a certain payment. The latter is equal to the certainty equivalent (CE):

$$CE_{\theta} = E(\pi_{\theta}) - RP_{\theta} \quad (12)$$

The risk premium can be approximated as follows (Pratt 1964):

$$RP_j \approx \frac{1}{2} r \text{Var}(\pi_{\theta}) \quad (13)$$

where r is the Arrow-Pratt risk coefficient of risk aversion, with the properties $r = -U''/U'$ and, for a risk-averse decision maker, $r > 0$. Farmers chose the optimal species diversity level, v^* , to maximize their certainty equivalent, $\max_v CE_{\theta}(k, v)$. This results in the following optimality condition (Baumgärtner 2007):

$$\frac{\partial E(\pi_{\theta}(v^*)) - (RP_{\theta}(v^*))}{\partial v} = \frac{\partial (C_{\theta}(v^*))}{\partial v} \quad (14)$$

Farmers chose species diversity at a level where the marginal certainty equivalent is equal to the marginal cost of species diversity provision. The optimal level of species diversity, v^* , increases with farmers’ degree of risk aversion, r , i.e. $\frac{\partial v^*}{\partial r} > 0$ (see also Baumgärtner 2007).

4.3 Theoretical Analysis of the Economic Model

In the theoretical analysis, we focus on implications of increasing drought risks for the optimal species diversity level and for the certainty equivalent. In detail we examine the following three aspects: i) impact of increasing drought risks, ii) species diversity to manage increasing droughts risks and iii) different implications for net sellers and net buyers considering the previous two aspects.

Impact of Increasing Drought Risks:

Lemma 1. *Increasing drought risks decrease farmer's certainty equivalent:*

$$\frac{\partial CE_{\theta}}{\partial k} < 0 \quad (15)$$

(Proof 1, Appendix 1)

Droughts decrease the expected hay yields while increasing their variance, i.e. the production risk, and droughts increase expected hay prices while increasing their variance, i.e. the price risk, as we outlined above. Given that the natural hedge is imperfect, droughts reduce farmers expected income and increase their income variance, i.e. the income risk. Consequently, increasing drought risks decrease the certainty equivalent. This holds for net sellers and net buyers (Proof 1).

Species Diversity to Manage Increasing Drought Risks:

Proposition 1. *Increasing drought risks increase the marginal certainty equivalent of species diversity and thus farmers' optimal level of species diversity, v^* :*

$$\frac{\partial^2 CE_{\theta}}{\partial k \partial v} > 0 \quad (16)$$

$$\frac{\partial v^*}{\partial k} > 0 \quad (17)$$

(Proof 2, Appendix 1).

We outline above that species diversity can mitigate negative impacts from droughts, as species diversity increases the expected hay yields and decreases their production risk.²³ Hence, it is beneficial for farmers to respond to a higher drought probability with higher levels of species diversity. The response is depending on the level of risk aversion of farmers (Proof 2).

Different Implications for Net Sellers and Net Buyers:

The adverse income effects of increasing drought risks are higher for net buyers than for net sellers, as net buyers do not benefit from the natural hedge. Actually, net buyers even have a disadvantage from the natural hedge, as they need to buy hay at a high price when hay yields are lower due to a

²³ Note that we assume that species diversity does not influence hay prices as only some farmers use species diversity for farm management. If such response would be created, the impact of species diversity would reduce, i.e. we would find less increase in the marginal certainty equivalent of species diversity and optimal species diversity level, however, given the natural hedge the here described effects would remain.

drought. Consequently, the farm types differ in their strategy to manage increasing drought risks with respect to species diversity:

Proposition 2. *Increasing drought risks increase the marginal certainty equivalent of species diversity and the optimal species diversity levels, v^* , more for net buyers than for net sellers:*

$$\frac{\partial^2 CE_{net\ buyer}}{\partial k \partial v} > \frac{\partial^2 CE_{net\ seller}}{\partial k \partial v} > 0 \quad (18)$$

$$\frac{\partial v_{net\ buyer}^*}{\partial k} > \frac{\partial v_{net\ seller}^*}{\partial k} > 0 \quad (19)$$

(Proof 3, Appendix 1).

Under increasing drought risks the change in the marginal effects of species diversity on income variance are higher for net buyers than for net sellers. Thus, under increasing drought risks the marginal diversity effects on expected utility and the optimal species diversity level are higher for net buyers than for net sellers (Proof 3).

4.4 Empirical Framework & Data

Next, we aim to quantify and empirically test the relationships discussed in the economic model, using a combination of stochastic production function (hay yields), structural vector autoregressive model (hay prices) and simulation (expected income, risk premium and certainty equivalent). The description of the analysis is ordered in four steps: First, we provide insights into the data to provide reasons for model selection and choices of simulation parameters. Second, we quantify effects of droughts and species diversity on hay yields and their variability. Third, we empirically assess drought effects on hay prices. Fourth, we combine all components to simulate implications of species diversity choices under different climatic conditions, farm types and farmers' levels of risk aversion.

4.4.1 Data

Hay Yield Data:

We use annual hay yield data from three coordinated drought biodiversity experiments (two Swiss sites, *site 1* and *site 2*, and one Irish site, *site 3*; see Hofer et al. 2016 and Haughey et al. 2018 for details²⁴). The experiments comprised four different monocultures and eleven different mixtures and hay yield information of treated (simulated summer drought²⁵) and untreated (control) plots. The

²⁴ Available hay yield data of the sites cover two, one and three years for site 1, site 2 and site 3, respectively, and hay was harvested six times at site 1 and five times at the other sites.

²⁵ Droughts were simulated by using tunnel-shaped rain-out shelters at randomly selected plots within a site (Haughey et al. 2018)

droughts simulated in the experiments represent ‘severe’ to ‘extreme’ summer droughts (Hofer et al. 2016). The species mixtures used in the experiments vary in species number and evenness (composition 1: 0.5, 0.5, 0, 0, composition 2: 0.79, 0.07, 0.07, 0.07, and composition 3: 0.25, 0.25, 0.25, 0.25). We use the Shannon-index as measure for diversity ($H = -\sum_j^N \Psi_j \ln(\Psi_j)$, where Ψ_j is the sown share of the species i, i, \dots, N , in a plot; Krebs 1999). The Shannon-index levels are: 0, 0.69, 0.74 and 1.39 (monoculture to most diverse mixture; see Table A2 for summary statistics).

Hay Price Data:

The hay prices are the bi-weekly reported average wholesale prices ex-farm including value added tax for high-pressure pressed hay in South Germany (including Bavaria and Baden-Württemberg) in Euro 100 kg⁻¹ and were provided by the Bavarian Association of Farmers (Fig. 1; Schaub and Finger 2019; Schaub and Finger 2020). The hay price data covers the period from August 2002 to April 2019 and are rather unique and of high quality as hay price data are seldom reported over such long time spans and with such a frequency. We transform hay prices into monthly natural logarithm real hay prices using the harmonized index of consumer prices for Germany with the base year 2015 taken from Eurostat (Eurostat 2019).

Weather Data:

For identifying drought events in our hay price time series we use the Standardized Precipitation Evapotranspiration Index (SPEI) as drought index, which contains precipitation and potential evapotranspiration information (Vicente-Serrano et al. 2010). We employ the SPEI and a threshold representing ‘severe droughts’ (Yu et al. 2014) to identify our drought variable, γ , during summer (July and August). We use severe summer drought to relate to the simulated droughts in the biodiversity experiments (see Appendix 2). We employ monthly 1 km x 1 km gridded data of potential evapotranspiration and precipitation for the period from January 1991 to March 2019, provided by the German Meteorological Service (DWD 2019), for computing the SPEI for each grid.²⁶ We then take the average of the SPEI across South Germany and identify, based on the SPEI and our threshold, ‘severe droughts’ in the summer months (Fig. 1).

Within the 17 years of our weather time series, we observe three drought years, hence, a drought probability of 3/17 (18%). We use this drought probability for defining the drought probability of the baseline scenario in the simulation.

²⁶ We use the R-package ‘SPEI’ for computing the SPEI (Beguería and Vicente-Serrano 2013).

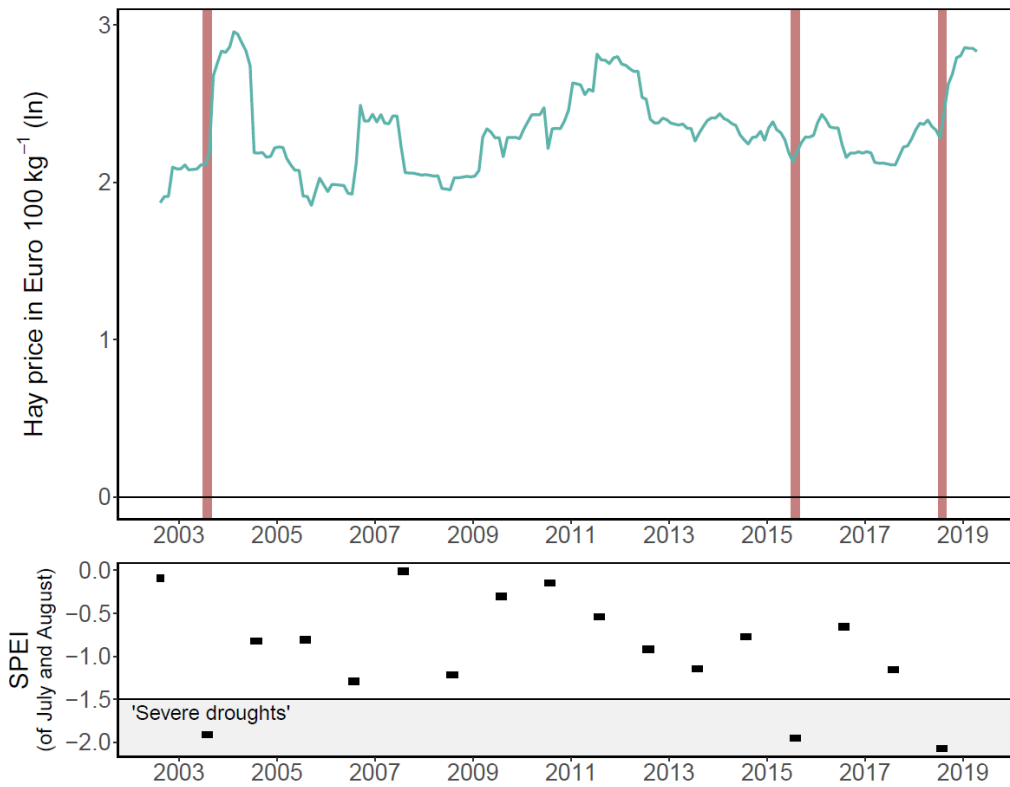


Figure 1: Hay prices, SPEI and ‘severe droughts’ in summer from August 2002 to March 2019 in South Germany. The upper panel shows hay prices (green line) and years with ‘severe droughts’ in summer (γ ; red bars). The lower panel shows the mean SPEI for the summer months (July and August; black dashes). The mean (variance) of the ln-transformed hay prices is 2.32 (0.06). The mean (variance) of the hay prices is 10.50 (8.31).

4.4.2 Econometric Implementation

Hay Yields:

We use a stochastic production function framework (Just and Pope 1978) to identify the drought effects hay yields and their variance as well as the diversity effect on hay yields and their variance under drought and non-drought conditions (see e.g. Finger and Buchmann 2015, Schaub et al. 2020a). We estimate the drought and the diversity effects on expected hay yields as:

$$y_{hay,i} = \beta_0 + \beta_\gamma \gamma_i + \beta_{\varphi v} \varphi_i v_i^{0.5} + \beta_{\gamma v} \gamma_i v_i^{0.5} + \beta_x Z_i + e_{1,i} \quad (20)$$

Where $y_{hay,i}$ is the annual hay yields²⁷ of plot i , β_0 is the intercept, and γ_i ²⁸ indicates droughts. $\varphi_i v_i^{0.5}$ represents the diversity effects in the control plots and $\gamma_i v_i^{0.5}$ the diversity effects in the drought plots. We use a square root specification of species diversity, reflecting possible saturation effects (see e.g. Schaub et al. 2020a). Moreover, this specification performs best compared to other specifications

²⁷ Annual yields refer to the period from the first cut of year to the first cut of the next year.

²⁸ Note that γ indicates a drought while k indicates increasing drought risks.

based on the Akaike information criterion.²⁹ Z_i is a set of control variables. The error term, $e_{1,i}$ includes all uncontrolled factors and is assumed to have an expected value of zero. The error term is used to quantify the variance of hay yields as follows $Var(y_{hay,i}) = (y_{hay,i} - \bar{y}_{hay,i})^2 = e_{1,i}^2$. Next, we estimate the effects of droughts and species diversity on variance of hay yields as:

$$Var(y_{hay,i}) = \beta_0 + \beta_{\varphi v} \varphi_i v_i^{0.5} + \beta_{\gamma v} \gamma_i v_i^{0.5} + \beta_X Z_i + e_{2,i} \quad (21)$$

We use robust standard errors to correct for heteroscedasticity in all estimations with species diversity.³⁰

Hay Prices:

We employ a structural vector autoregressive model of type A (SVAR) to identify the drought effects (exogenous shock) on hay prices (endogenous variables; see e.g. Lütkepohl 2005).³¹ Our SVAR is defined as:

$$AX_t = \mu_t + A_1^* X_{t-1} + \dots + A_d^* X_{t-d} + B \varepsilon_t \quad (22)$$

Where t represents time and d the maximal lag length in the model ($t = 1, \dots, T, T = MG$, where M is number of months in a year and G number of years). The vector X_t ($K \times 1$) includes drought events, γ , and hay prices, p_{hay} . μ_t is a constant. The dimension of the coefficient matrices A_j^* for $j = 1, \dots, d$ is $K \times K$. B is an identity matrix, and the structural error, ε_t , is assumed to be white noise. By multiplying equation 22 with A^{-1} we obtain:

$$X_t = A^{-1} A_1^* X_{t-1} + \dots + A^{-1} A_d^* X_{t-d} + A^{-1} B \varepsilon_t \quad (23)$$

Where $u_t = A^{-1} B \varepsilon_t$ is the reduced form residuals vector and $\sum_u A^{-1} B B' A^{-1}$ is its variance-covariance matrix. For identification of our structural model, we impose a restriction on the reduced form residuals vector, i.e. droughts affect hay prices instantaneously but hay prices do not affect droughts:

$$\begin{bmatrix} u_t^\gamma \\ u_t^{hay} \end{bmatrix} = \begin{bmatrix} 1 & 0 \\ \alpha_{hay,\gamma} & 1 \end{bmatrix} \begin{bmatrix} \varepsilon_t^\gamma \\ \varepsilon_t^{hay} \end{bmatrix} \quad (24)$$

We select the optimal lag length of the model based on hay prices using the Akaike information criterion and we test for stationarity using the Augmented Dicky Fueller. We are here primarily interested in the annual effects of droughts, i.e. the drought from the first cut during a drought year

²⁹ Other specifications include: v_i , $v_i + v_i^2$, $\ln(v_i)$.

³⁰ For robust estimation we used the R-package 'estimatr' (Blair et al. 2018).

³¹ See also Schaub and Finger (2020). For analysing the price data, we used the R-packages 'vars' and 'urca' (Pfaff 2008a, Pfaff 2008b).

to the first cut of the next year (annual drought price effect). Therefore, from the SVAR's impulse response function, we derive the mean drought effect (and confidence interval) on hay prices for the months from the drought event to the first cut in the following year, M_1 . In the months before the drought, M_0 , the drought effects are zero, thus:

$$\text{annual drought price effect} = (\sum_{m=1}^{M_1} \text{drought price effect}_m) / M \quad (25)$$

Where $m = 1, \dots, M_1$. Additionally, we follow equation 25 for deriving the upper and lower confidence intervals. From the confidence intervals we compute the standard deviation of the estimate ($sd(\text{annual drought price effect}) = ((\text{upper confidence interval} - \text{lower confidence interval}) / (2 \cdot 1.96))^{0.5}$; see e.g. Greene 2012).

4.5 Simulation

Next, we simulate effects of species diversity and droughts using empirical estimates derived in the previously described steps. We simulate the effects of different climatic conditions, by varying drought probabilities, considering differences in farm types (i.e. net position) and farmers' risk aversion (Table 1). For the farm types, we focus on two distinct types in the main analysis: net sellers with a net position of 0% and net buyers with a net position of 130%. The net position indicates the relatively amount of farmers' hay demand compared to the farmers' hay yields (hay demand/expected hay yields 100). Furthermore, we employ average values for farm types and market condition (Table A1).³²

Table 1: Overview of the simulation specification.

Drought probability of the scenarios	Net position (hay demand/expected hay yields 100)	Risk aversion
Drought probability 0% ^{sa}	Net seller (0%)	'Risk neutral' ($r = 0$) ^{sa}
Drought probability 18% (Baseline)	Net buyer (130%)	'Low risk averse' ($r = 0.5$) ^{sa}
Drought probability 28% ^{sa}	From 0% to 200% ^{sa}	'Rather risk averse' ($r = 2$)
Drought probability 38%		'Very risk averse' ($r = 4$) ^{sa}
Drought probability 48% ^{sa}		
Drought probability 100% ^{sa}		

Note: We analyze all different combination of drought probabilities with net positions and with risk aversions. The drought probability of the baseline scenario is based on the observed droughts during the price time series. Gray shaded cells = setting of the main analysis. sa = part of the sensitivity analysis. Source: Hardaker et al. 2015, NCCS 2018, Spinoni et al. 2018, Grillakis 2019, Iyer et al. 2020.

³² We choose average values from South Germany, as our hay price data are from this region and results from experimental yield data from Switzerland and Ireland are better transferable than vice versa.

We run the simulation over 25 years using the different drought probabilities. In each simulation year we draw hay yields and hay prices based on their means and variances³³ and add in drought years the drought effects to the hay yields and hay prices. Based on our theoretical model, we are using a bivariate normal distribution for simulating hay yields and hay prices (see Appendix 2).³⁴ For hay yields, we additionally consider different levels of species diversity. Next, we calculate the expected value and variance of hay yields and prices as well as of revenues and income of net sellers and net buyers over the 25 years of the simulation. Based on these results we compute the certainty equivalent and the risk premium. We repeat this procedure 100 times for each scenario. Furthermore, we analyze the simulation output using regression analysis (see Appendix 2) and we test the differences between effects using a Wald-test-based comparison (Fox and Weisberg 2019).

4.6 Empirical Results

4.6.1 Hay Yields and Hay Prices

Our econometric analysis using experimental data on hay yields (Panel A of Fig 3), show that severe (and extreme) summer droughts led on average to annual hay yield reductions of about $-900 \text{ kg ha}^{-1} \text{ year}^{-1}$ (-8%; Fig. 2). Moreover, species diversity increased hay yields under both drought and non-drought conditions. The diversity effects in grassland plots with and without droughts were not significantly different from each other (based on Wald-test-based comparison³⁵, $\chi^2 = 1.38$). Species diversity decreased variance of hay yields. The effects were not significantly different under drought and under non-drought conditions ($\chi^2 = 0.68$). Analyzing hay price data, we find that the average drought effects on hay prices were about +11% (Panel B of Fig. 2).³⁶

³³ Note that before running the simulation we removed the 12 month drought effects from the hay price mean and variance.

³⁴ Using copula distributions, even though they might be theoretically favored, is not feasible due to data constrains.

³⁵ See Fox and Weisberg (2019) for details.

³⁶ The hay price and drought time series were stationary (Table A3) and the suggested lag length of the hay prices by the Akaike information criterion was two.

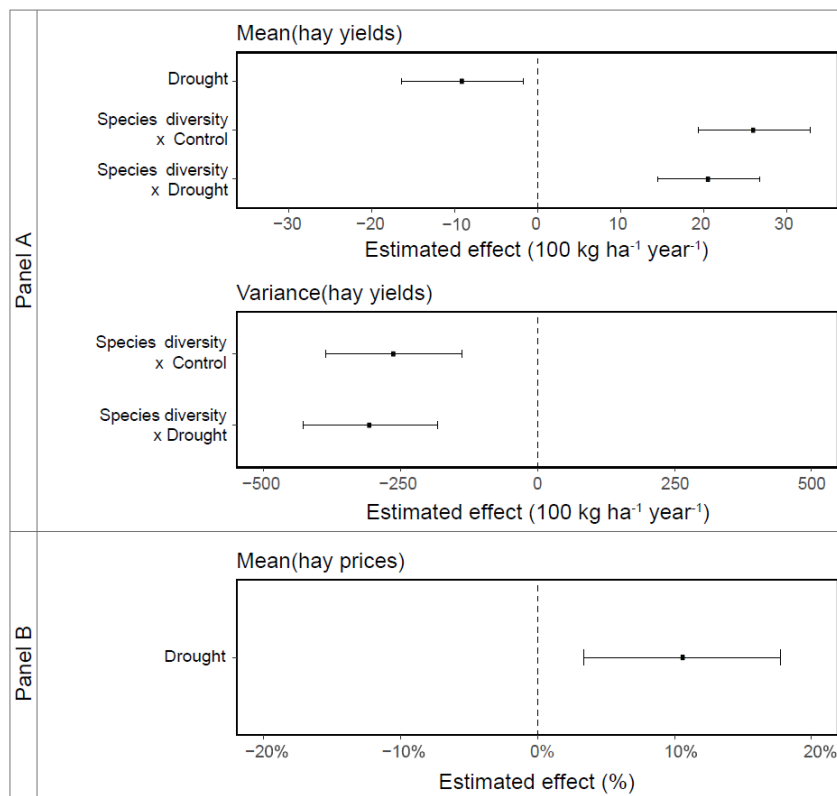


Figure 2: Drought and diversity effects on hay yields and their variance (100 kg ha⁻¹ year⁻¹) as well as on hay prices (%).³⁷ Drought indicates $\gamma_i = 1$ and control indicates $\varphi_i = 1$. The interaction of species diversity ($v_i^{0.5}$) with control or drought are the coefficient estimates of $\beta_{\varphi v}$ and $\beta_{\gamma v}$ (see equation 20 and 21). The error bars indicate the 95%-confidence interval. Mean(hay yields) intercept = 123.73. Variance (hay yields) intercept = 787.12. The full regression results are available in Table A5.

4.6.2 Simulation Results

Main Results:

Our simulation results show that increasing drought risks, i.e. comparing the drought probability 38% to drought probability 18% (baseline) scenario, did not affect net sellers' (0%) expected income, risk premium or certainty equivalent.³⁸ This was due to a strong natural hedge in drought years (Fig. 3, Table 2).

Moreover, species diversity increased net sellers' expected income and decreased their risk premium. Therefore, species diversity increased the certainty equivalent. The diversity effects on net sellers'

³⁷ For calculating the drought effects we followed equation 25, with $M = 12$, $M_1 = 9$ (Hofer et al. 2016; see Appendix 2 for calculation).

³⁸ For net sellers, the simulated expected income, risk premium and certainty equivalent were 18669, 1125 and 17544 Euro farm⁻¹, respectively, considering the average monoculture in the drought probability 18% (baseline) scenario.

expected income, risk premium and certainty equivalent did not alter with increasing drought risks (Table 2). For quantifying the diversity effects, we compare the diversity effect size at the average species diversity level of all mixtures in the experiment (Shannon-index = 0.80) to the average monoculture, henceforth called the 'average diversity effects'. The average diversity effects on the expected income, risk premium and certainty equivalent in the drought probability 18% (baseline) and drought probability 38% scenario were +3971, -331, +4302 Euro farm⁻¹ (+21%, -29%, +25%), respectively.

Net buyers' (130%) expected income decreased and risk premium increased with increasing drought risks by -604 and +80 Euro farm⁻¹ (-1% and +11%; Fig. 3, Table 2), respectively, considering the average monoculture.³⁹ Thus, increasing drought risks reduced the certainty equivalent by -684 Euro farm⁻¹ (-1%). Note that the income reduction in a single drought year was -2501 Euro farm⁻¹ (-4%) for net buyers (Table A11).

Species diversity increased the expected income and decreased the risk premium of net buyers, leading to an increasing certainty equivalent with increasing species diversity. Increasing drought risks did not alter these diversity effects. The average diversity effects on the expected income, risk premium and certainty equivalent in the drought probability 18% and drought probability 38% scenario were +3971, -287 and +4258 Euro farm⁻¹ (+7%, -39%, +8%), respectively.

Furthermore, the diversity effects on expected income were identical for net sellers and net buyers in each scenario (Table A9). The diversity effects on risk premium as well as on certainty equivalents were slightly, but not significantly, higher for net sellers than for net buyers. This was because the positive drought effects on hay prices compensated for the negative drought effects on hay yields (Table A10).

When considering instead of the estimated price response to drought lower prices responses, we find that increasing drought risks also decreases expected income and increases risk premium of net sellers, thus, decreases their certainty equivalent (Fig. A1 and A2).⁴⁰ These changes in hay price responses to droughts affected net sellers more than net buyers.

³⁹ For net buyers, the simulated expected income, risk premium and certainty equivalent were 55940, 727 and 55123 Euro farm⁻¹, respectively, considering the average monoculture in the drought probability 18% (baseline) scenario.

⁴⁰ We consider lower price responses as previous literature often identified an 'imperfect' (see e.g. Finger 2012, Ramsey et al. 2019).

Sensitivity Analysis 1 – Different Drought Probabilities:

In this sensitivity analysis, we compared the baseline scenario to a wider range of scenarios, i.e. drought probabilities (drought probability 0%, drought probability 28%, drought probability 48% and drought probability 100%).

Net sellers' expected income did not change with changes in drought probabilities, whereas the risk premium in the drought probability 0%, 48% and 100% scenario compared to the drought probability 18% (baseline) scenario changed by -110, 78, 162, Euro farm⁻¹, respectively, considering the average monoculture (Fig. 3, Table A9 and A10). Yet, net sellers' certainty equivalents did not alter with different drought probabilities, as the effects on expected income and risk premium canceled each other out. The diversity effects on net sellers' expected income, risk premium and certainty equivalent were always positive and large. In general, the diversity effects did not change with changes in drought probability compared to the baseline scenario. This holds with few exceptions: the diversity effects on expected income decreased in the drought probability 100% scenario as well as the diversity effects additionally decreased the risk premium in the drought probability 48% and drought probability 100% scenario compared to the drought probability 18% (baseline) scenario.

Increasing drought risks decreased expected income and increased the risk premium of net buyers. Thus, increasing drought risks decreased net buyers' certainty equivalent (Table A9 and A10). Only in the drought probability 0% scenario the expected income did not change compared to the drought probability 18% (baseline) scenario. The species diversity strongly increased the expected income and certainty equivalent and decreased the risk premium independent of the drought probability. These effects were generally independent of the drought probability. Only in the drought probability 100% scenario the diversity effects were less increasing on the expected income and more increasing on the risk premium of net buyers than in drought probability 18% (baseline) scenario.

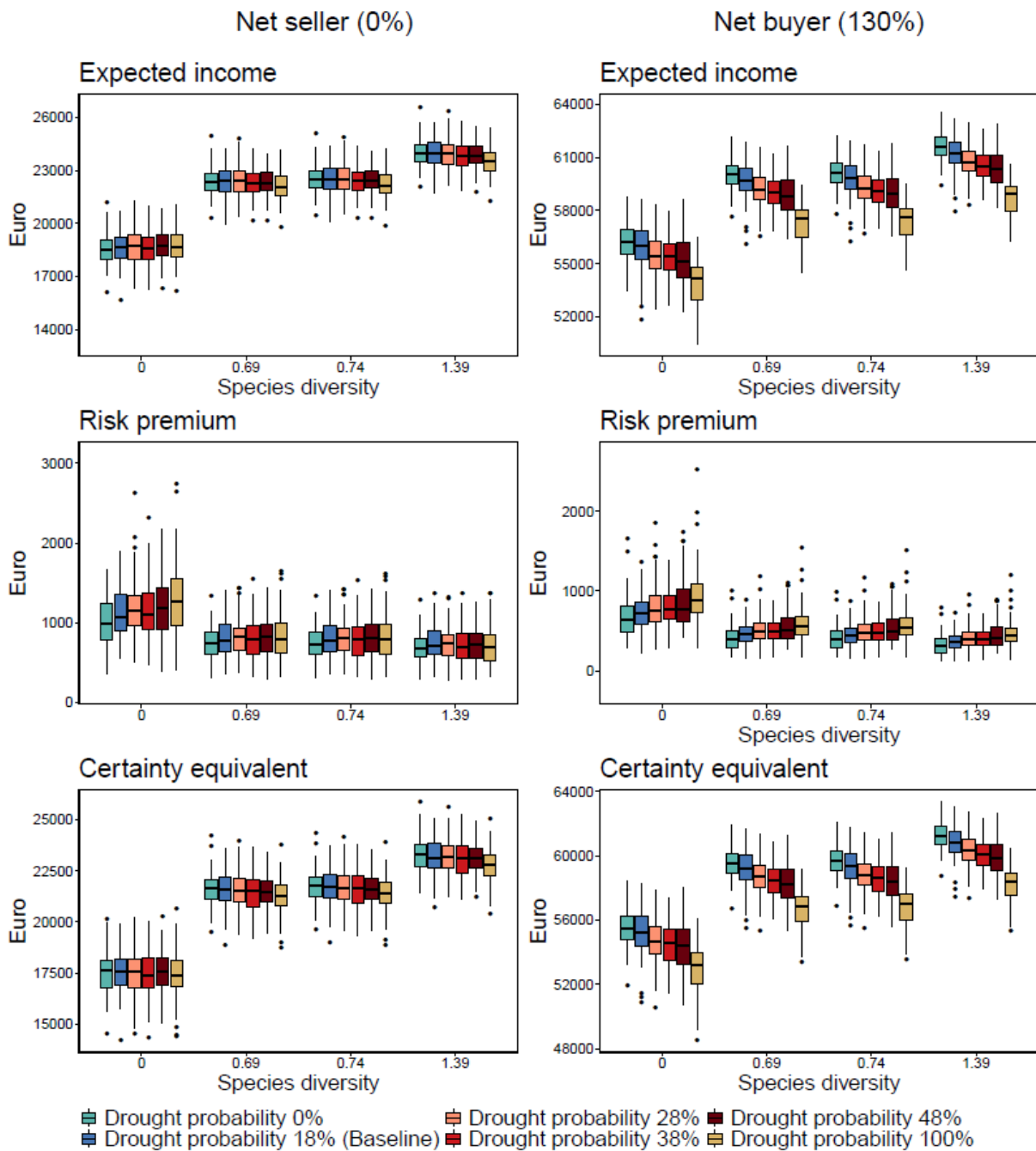


Figure 3: Effects of increasing drought risks (i.e. increasing drought probability) and species diversity on expected income, risk premium and certainty equivalent (in Euro farm⁻¹ year⁻¹). The Arrow-Pratt risk coefficient of risk aversion, r , was assumed to be 2. Part of the main analysis are the drought probability 18% (baseline) and drought probability 38% scenarios.

Note: We assume a farm size of 16.5 ha, which is based on the average farm size in South Germany (Table A1). In the drought probability 18% (baseline) scenario considering the average monoculture the mean net sellers' expected income (risk premium) was 18669 (1125) Euro farm⁻¹ year⁻¹. The respective value for the average net buyers was 55940 (727) Euro farm⁻¹ year⁻¹. All farms are assumed to receive an additional income of 20000 Euro farm⁻¹. See Table A1 and A6 for calculation basis.

Table 2: Marginal effects of increasing drought risks and species diversity on net sellers' and net buyers' expected income, risk premium and certainty equivalent and differences in marginal effects between net sellers and net buyers.

Effect	Farm type	Expected income	Risk premium	Certainty equivalent
Effect of increasing drought risks	Net seller	-46.78 (-0.38)	44.54 (0.91)	-91.32 (0.44)
	Net buyer	-603.91 (13.47)***	80.49 (5.99)**	-684.4 (13.86)***
Diversity effects	Net seller	4483.4 (43.25)***	-333.10 (-9.82)***	4816.50 (44.24)***
	Net buyer	4483.4 (34.55)***	-305.63 (-12.66)***	4789.03 (33.59)***
	Difference	0 (0)	27.47 (0.44)	-27.47 (0.02)
Additional diversity effects under increasing drought risks	Net seller	-86.42 (0.37)	-73.88 (2.09)	-12.54 (0.01)
	Net buyer	-86.42 (0.22)	-30.26 (0.74)	-56.17 (0.08)
	Difference	0 (0)	27.59 (0.20)	-27.59 (0.01)

Note: We compare in this table the drought probability 38% scenario to the drought probability 18% (baseline) scenario. The Arrow-Pratt risk coefficient of risk aversion, r , was assumed to be 2. Numbers in parentheses are t-values or in the case of cumulative effects they are χ^2 . χ^2 values are from a Wald-test-based comparison (Fox and Weisberg 2019). The effect sizes are from the regression results (Table A7-A8). * $p < 0.1$; ** $p < 0.05$; *** $p < 0.01$. Additional results are in Table A9-A10.

Sensitivity Analysis 2 – Different Net Positions:

When considering a wider range of net position instead of the so far considered two farm types, we found that farmers' net position greatly influenced the effects of increasing drought risks on expected income, variance and certainty equivalent (Fig. 4). However, while the net position greatly influenced the variance, the effect on the risk premium was relatively low due to substantial expected income increase with net position, thus, with milk sold.

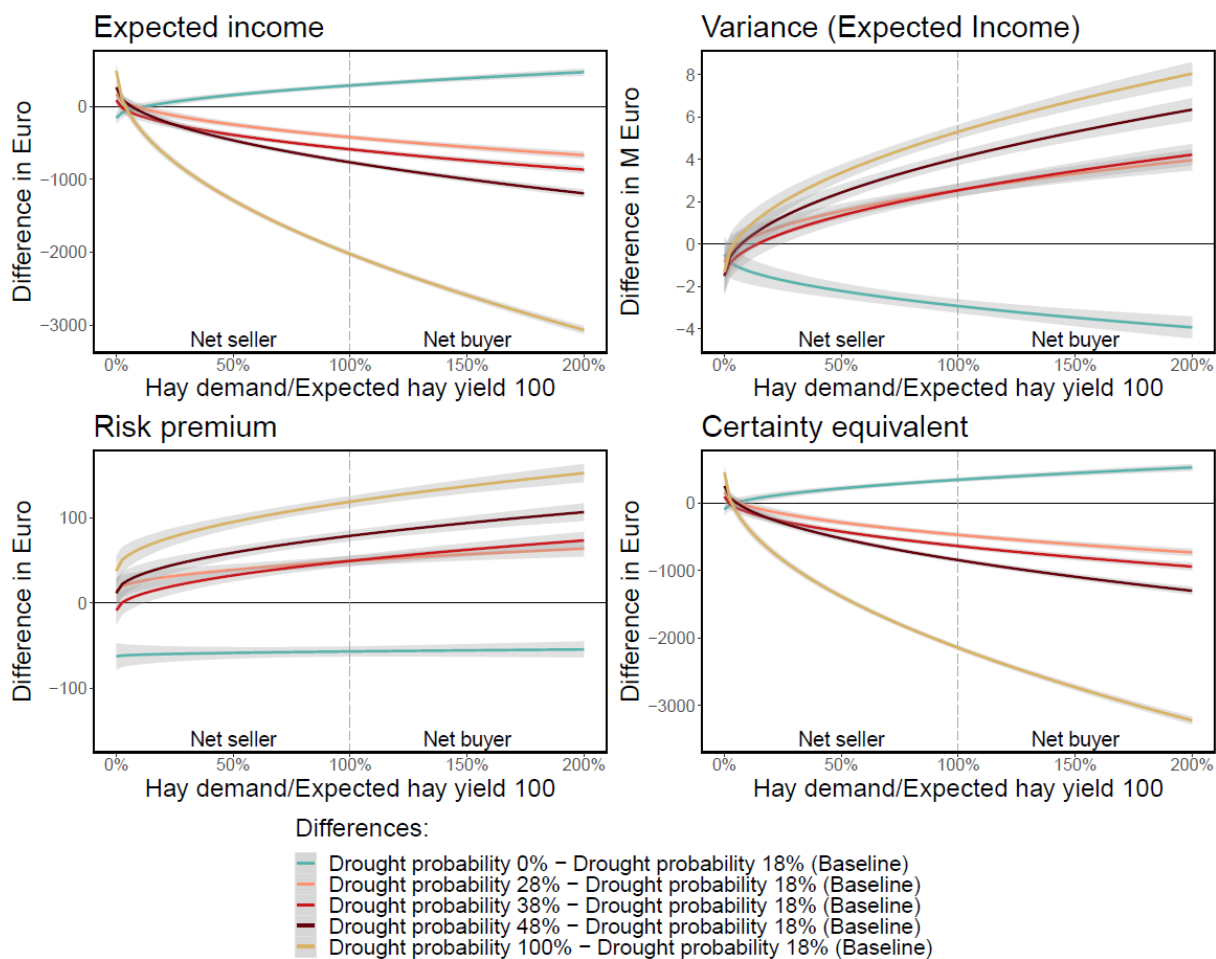


Figure 4: Changes in increasing drought risk effects with farmers' net position (hay demand/expected hay yields 100) for expected income, risk premium and certainty equivalent in Euro farm⁻¹ year⁻¹ and for variance in Million Euro farm⁻¹ year⁻¹. The figures shows values for the average species diversity level (Shannon-index = 0.80) and a coefficient of risk aversion of 2. *Note:* The average farm size is 16.5 ha and all farms are assumed to receive an additional income of 20000 Euro farm⁻¹ (Table A1).

Sensitivity Analysis 3 – Different Levels of Risk Aversion:

Here, we consider different levels of risk aversion (next to 'rather risk averse' ($r = 2$) also 'low risk averse' ($r = 0.5$) and 'very risk averse' ($r = 4$) farmers). The risk premium for net sellers and net buyers differed greatly by risk aversion (Fig. 5).⁴¹ Also the effect sizes of the increasing drought risks and diversity effects changed with risk aversion. For example, net buyers risk premium increased with increasing drought risks (from the drought probability 18% (baseline) to the drought probability 38% scenario) by 20, 80 and 161 Euro farm⁻¹ for low, rather and very risk averse farmers, respectively. The direction of the effects of increasing drought risks (i.e. increasing) and species diversity (i.e. decreasing)

⁴¹ Net sellers' risk premiums at a certain level of risk aversion were lower than net buyers' risk premiums due to higher expected income and despite a higher income variance of net sellers (Table A1).

on the risk premium were the same for rather risk averse as well as less and more risk averse net sellers and net buyers (Table A10).

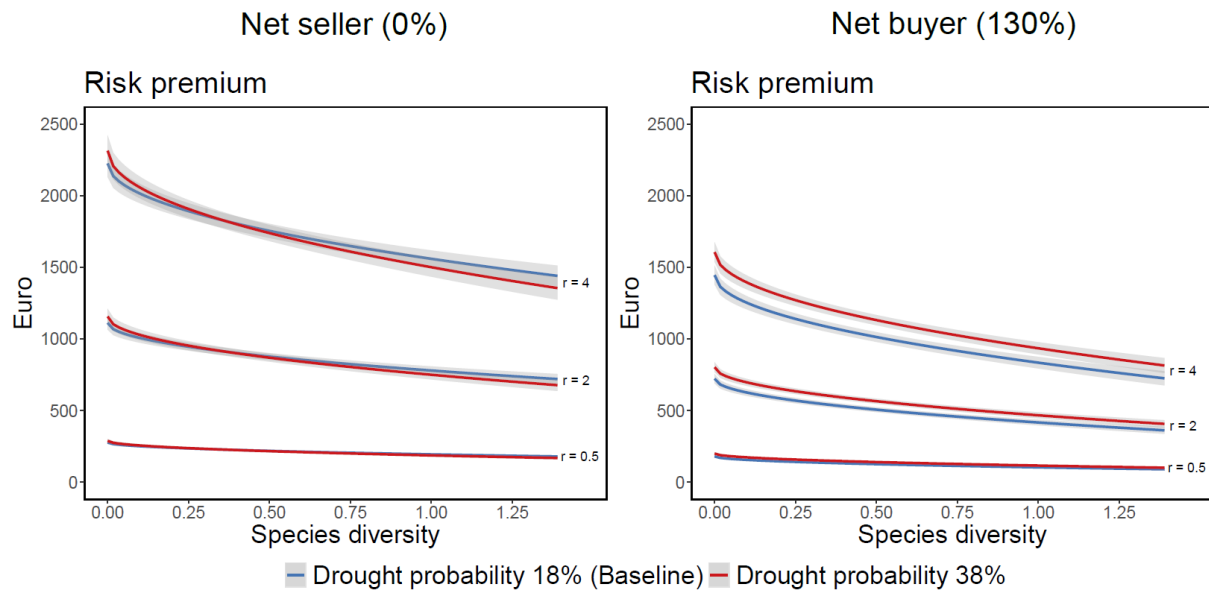


Figure 5: Increasing drought risks and diversity effects on net sellers' and buyers' risk premium Euro farm⁻¹ year⁻¹ over a range of risk aversion coefficients. *Note:* The average farm size is 16.5 ha and all farms are assumed to receive an additional income of 20000 Euro farm⁻¹ (Table A1). See Fig. A3 for increasing drought risks and diversity effects on net sellers' and net buyers' variance of expected income.

4.7 Discussion & Conclusion

We analyze the effects of increasing drought risks on farmers as well as if and how species diversity can be used to mitigate adverse effects of increasing drought risks. In the analysis we explicitly address implication of farmers' risk preferences, different farm types and market response to drought shocks using theoretical and empirical analyses.

We address in our analyses three aspects in detail: First, we show in our theoretical analysis, which is based on previous research, that increasing drought risks decreases farmer's certainty equivalent (*Lemma 1*). Our empirical analysis confirms the theoretical analysis for net buyers and almost all net sellers. Yet, increasing drought risks did not adversely affect net sellers that directly sold all or almost all of their produced hay. This was because drought induced sharp hay price increases balanced hay yield losses in our sample. When we considered lower drought price response (in line with an 'imperfect' natural hedge), increasing drought risks negatively affected all farmers. Adverse increasing drought risk effects for farmers decreased generally with lower net positions. For farmers that were negatively affected by increasing drought risks, the decrease in expected income mostly caused the decrease in certainty equivalent, while the effects on the risk premium were small in absolute terms.

The relatively low drought effects in our sample on annual hay yields (-8%) were due to yield-overcompensation in post-drought harvests (Finn et al. 2018). In general, previous research often found negative drought effects on annual hay yields. These effects can be heterogeneous and are depending on site-specific climatic and soil conditions as well as overcompensation in post-drought harvests (Finger et al. 2013, Craven et al. 2016, Hofer et al. 2016, Wagg et al. 2017). Moreover, previous research showed that regional droughts substantially affected hay prices (while not affecting other feed prices) and linked these results to low market integration of hay markets (Schaub and Finger 2020).

Second, in our theoretical analysis we show that increasing drought risks increase the marginal certainty equivalent of species diversity and thus farmers' optimal level of species diversity (*Proposition 1*). This holds for farmers independent of their risk aversion but effects increase with risk aversion. The empirical analysis shows that species diversity increased farmers' certainty equivalent – as it increased the expected income and decreased the risk premium. These diversity effects were, in contrast to the theoretical analysis, overall independent from drought probability. The general positive diversity effects are in line with previous economic studies (e.g. Finger and Buchmann 2015, Binder et al. 2018, Schaub et al. 2020a, Schaub et al. 2020b). The findings also support previous studies that showed for several experiments positive diversity effects, which did not alter under drought treatment (Craven et al. 2016, Hofer et al. 2016). However, under different climate condition, soil condition and species diversity gradient additional positive diversity effects in drought years on hay yields and their stability might still exist, as other studies showed positive diversity effects on drought resistance, resilience and/or recovery in grasslands (Van Ruijven and Berendse 2010, Vogel et al. 2012, Isbell et al. 2015, Wagg et al. 2017). Furthermore, Haughey et al. (2018) showed, at the same experimental sites as in our study, that intra-annual stability increased with species diversity, which can also be valuable for farmers.

Third, the theoretical analysis revealed that increasing drought risks decrease the certainty equivalent more for net buyers' than for net sellers. Moreover, it shows that increasing drought risks increase the marginal certainty equivalent of species diversity more for net buyers than for net sellers, because of the diversity effects on the risk premium (*Proposition 2*). These differences between farm types are because net sellers benefit from natural hedge whereas net buyers need to purchase hay at a higher hay price. We find empirical evidence that the impact of increasing drought risks was higher for net buyers than for net sellers but we cannot empirically confirm that the marginal certainty equivalent of species diversity were different between farm types. Indeed, because the drought effects on hay prices over-compensated the drought effects on hay yields in our sample we even found a non-significant but small oppositional impact on the marginal certainty equivalent of species diversity to the one

proposed in our theoretical analysis. Overall, we find only small effects of increasing drought risks and species diversity on the risk premium. Therefore, our empirical analysis suggest that when the theoretical results between farm types only differ due to effects on the risk premium that these differences are most likely rather negligible.

In our economic model we assume that farmers can purchase hay from somewhere, however, when no hay or other appropriate substitutes⁴² are available droughts might lead for net buyers even to emergency slaughtering, which would even lead to higher losses and extreme volatility (e.g. Finger et al. 2018). We focus in the analysis on hay yields and hay prices. If we would consider that milk prices would be positively affected by droughts the market structure, particularly the market integration, of the milk and the hay market suggest that we would most likely find effects with the same signs as outlined above. This is as the regional milk market is supplied by a large region (Fousekis et al. 2017, Zhang 2017) whereas the hay market is supplied by a small region (Rudstrom 2004, Schaub and Finger 2020). Thus, it is likely that milk prices increase less in response to droughts than hay prices. Hence, if the milk market incorporates a natural hedge it can be assumed to be lower than the one of the hay market.

Our findings have several relevant implications: First, species diversity should be considered in the sustainable management of grasslands. Especially, as even under increasing drought risks due to climate change species diversity can increase expected income and reduce income risk. Second, it is important that extensions services when consulting farmers and policy makers when designing agricultural policies to cope with climate change impacts consider the differences in vulnerability to increasing drought risks of the different farm types. We highlight that the differences in vulnerability hold also in other agricultural setting, thus, it is generally important to consider whether farmers sell directly their agricultural production (and to what share) or if it is further processed within the farm.

Farmers can sow or over-sow grasslands with already available mixtures and adapt management intensity to increase and maintain grassland species diversity (see e.g. Walker et al. 2004). Moreover, new technologies, such as precision farming, might offer sustainable and more cost efficiently practice to increase and maintain species diversity (Finger et al. 2019). Cost reduction in the provision of more species diverse grasslands would also increases in the optimal species diversity level. Additionally, species diversity can also promote other ecosystem functions and services (Potts et al. 2009, Nyfeler

⁴² Note that farmers are often constraint by substitution hay with other feed as they produce under certain schemes (see e.g. Mack and Huber 2017 or Paredes et al. 2018) or simply for other economic and agricultural reasons.

et al. 2011, Suter et al. 2015, Suter et al. 2017, Connolly et al. 2018, Le Clec'h et al. 2019) that are not included in our valuation but that also can create positive values for farmers as well as for other private and public actors.

Our analysis offers several interesting future extensions, including the analysis of i) the differences between the private and social optimal species diversity level given market price responses, risk aversion and differences in farm types, ii) the impact of policies and insurances on the private and social optimum and iii) if and how policies and insurances can be used to align the private and social optimum.

4.8 Data Availability

For some years and harvests the data is online available at Hofer et al. 2016 and Haughey et al. 2018. For other years and harvest the data is not online available.

4.9 Code Availability

All R-code of the data preparation, analysis and simulation will be available upon request (after publication).

4.10 Acknowledgements

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4.12 Appendix 1 – Theoretical Analysis

In the proofs of the theoretical analysis we focus for net sellers on the distinct case of net sellers that only sell hay but do not produce milk (net position = 0%). This simplifies and clarifies the proofs. However, the proposed mechanisms and effects of the lemma and the propositions can be shown for various farm types, thus net positions, along the same lines.

4.12.1 Detailed Variance Definition

Considering that i) hay yields and hay prices follow a bivariate normal distribution (see e.g. Burt and Finley 1968, Bohrnstedt and Goldberger 1969) and ii) the assumption of deterministic milk production and milk price levels, which are uncorrelated with hay production and not influenced by droughts, net sellers and net buyers' income variances are defined as:

$$\begin{aligned} Var(\pi_{net\ seller}) &= E(y_{hay})^2 Var(p_{hay}) + E(p_{hay})^2 Var(y_{hay}) + \\ &2 E(y_{hay}) E(p_{hay}) Cov(y_{hay}, p_{hay}) + Var(p_{hay}) Var(y_{hay}) + Cov(y_{hay}, p_{hay})^2 \end{aligned} \quad (A.1)$$

$$\begin{aligned} Var(\pi_{net\ buyer}) &= E(y_{hay})^2 Var(p_{hay}) + E(p_{hay})^2 Var(y_{hay}) + \\ &2 E(y_{hay}) E(p_{hay}) Cov(y_{hay}, p_{hay}) + Var(p_{hay}) Var(y_{hay}) + Cov(y_{hay}, p_{hay})^2 + \\ &d_{hay}^2 Var(p_{hay}) + E(p_{hay})^2 Var(d_{hay}) + 2 d_{hay} E(p_{hay}) Cov(d_{hay}, p_{hay}) + \\ &Var(p_{hay}) Var(d_{hay}) + Cov(d_{hay}, p_{hay})^2 + Cov(y_{hay} p_{hay}, d_{hay} p_{hay}) \\ &= E(y_{hay})^2 Var(p_{hay}) + E(p_{hay})^2 Var(y_{hay}) + 2 E(y_{hay}) E(p_{hay}) Cov(y_{hay}, p_{hay}) + \\ &Var(p_{hay}) Var(y_{hay}) + Cov(y_{hay}, p_{hay})^2 + d_{hay}^2 Var(p_{hay}) + Cov(y_{hay} p_{hay}, d_{hay} p_{hay}) \end{aligned} \quad (A.2)$$

Note that $Cov(y_{hay} p_{hay}, d_{hay} p_{hay})$ is negative and likely to be high.

4.12.2 Proofs

Proof 1. – Lemma 1. Increasing drought risks decrease farmer's certainty equivalent:

First, the drought effects on expected income (equation 2) are negative, i.e. $\frac{\partial E(\pi_{\theta})}{\partial k} < 0$, considering an 'imperfect' natural hedge as well as that costs, hay demand, milk production and prices are not affected by droughts. Second, droughts cause income variance of net sellers and net buyers (equation A.1 and A.2) to increase, as the variance of hay yields and hay prices increase, i.e. $\frac{\partial Var(\pi_{\theta})}{\partial k} > 0$.⁴³

Therefore, increasing drought risks decrease farmer's certainty equivalent:

⁴³ Variance of hay yields and hay prices are not perfectly negative correlated.

$$\frac{\partial CE_{\theta}}{\partial k} < 0 \quad (\text{A.3})$$

Proof 2. – Proposition 1. Increasing drought risks increase the marginal certainty equivalent of species diversity and thus farmers' optimal level of species diversity, v^ :*

The marginal effects of species diversity on expected revenues increase with increasing drought risks

$$\frac{\partial^2 E(\pi_{\theta})}{\partial k \partial v} = \frac{\partial^2 E(y_{hay})}{\partial k \partial v} > 0. \text{ This holds under the condition that there are no feedback effects of species}$$

diversity on hay price levels, i.e. $\frac{\partial^2 E(p_{hay})}{\partial k \partial v} = 0$. Moreover, under increasing drought risks the marginal

effects of species diversity on variance reduction increase $\frac{\partial^2 Var(\pi_{\theta})}{\partial k \partial v} = \frac{\partial^2 Var(y_{hay})}{\partial k \partial v} < 0$. Thus, the

marginal certainty equivalent of species diversity increases with increasing drought risks:

$$\frac{\partial^2 CE_{\theta}}{\partial k \partial v} > 0 \quad (\text{A.4})$$

Therefore, we can prove that increasing drought risks increase farmers' optimal species diversity level by using the implicit function theorem:

$$\frac{\partial v^*}{\partial k} = - \frac{\partial^2 CE_{\theta}}{\partial v \partial k} / \frac{\partial^2 CE_{\theta}}{\partial v^2} \quad (\text{A.5})$$

Moreover, from equation 5, 6 and 11 we can derive:

$$\frac{\partial^2 CE_{\theta}}{\partial v^2} < 0 \quad (\text{A.6})$$

Consequently, we can show that equation A.5 is positive and that increasing drought risks increase farmers' optimal level of species diversity:

$$\frac{\partial v^*}{\partial k} > 0 \quad (\text{A.7})$$

Proof 3. – Proposition 2. Increasing drought risks increase the marginal certainty equivalent of species diversity and the optimal species diversity levels, v^ , more for net buyers than for net sellers:*

First, the impact of increasing drought risks on the income of net sellers and net buyers can be expressed as:

$$\frac{\partial E(\pi_{net\ seller})}{\partial k} = \frac{\partial E(y_{hay})}{\partial k} E(p_{hay}) + \frac{\partial E(p_{hay})}{\partial k} E(y_{hay}) + \frac{\partial Cov(E(y_{hay}), E(p_{hay}))}{\partial k} \quad (\text{A.8})$$

$$\frac{\partial E(\pi_{net\ buyer})}{\partial k} = \frac{\partial E(y_{hay}) - \partial d_{hay}}{\partial k} E(p_{hay}) + \frac{\partial E(p_{hay})}{\partial k} (E(y_{hay}) - d_{hay}) + \frac{\partial Cov(E(y_{hay}) - d_{hay}, E(p_{hay}))}{\partial k} \quad (A.9)$$

Considering that the increasing drought risks impact on the covariance in equations A.8 and A.9 describes the joint variability under increasing drought risks of either $E(y_{hay})$ and $E(p_{hay})$ or $E(y_{hay}) - d_{hay}$ and $E(p_{hay})$, we can express their derivatives for net sellers and buyers as $\frac{\partial E(y_{hay})}{\partial k} \frac{\partial E(p_{hay})}{\partial k}$ and $\frac{\partial E(y_{hay}) - d_{hay}}{\partial k} \frac{\partial E(p_{hay})}{\partial k}$, respectively. These two expressions correspond to the natural hedge effect on income.

Considering equation A.8 and A.9, the marginal effects of species diversity on expected income under increasing drought risks are given by $\frac{\partial^2 E(y_{hay})}{\partial k \partial v} E(p_{hay})$. This holds under the condition that there are no feedback effects of species diversity on hay price levels, i.e. $\frac{\partial^2 E(p_{hay})}{\partial k \partial v} = 0$, and on hay demand, i.e. $\frac{\partial^2 d_{hay}}{\partial k \partial v} = 0$.⁴⁴

Second, the impacts of increasing drought risks on income variance of net sellers and net buyers are:

$$\frac{\partial Var(\pi_{net\ seller})}{\partial k} = \frac{\partial E(y_{hay})^2 Var(p_{hay}) + E(p_{hay})^2 Var(y_{hay}) + 2 E(y_{hay}) E(p_{hay}) Cov(y_{hay}, p_{hay}) + Var(p_{hay}) Var(y_{hay}) + Cov(y_{hay}, p_{hay})^2}{\partial k} \quad (A.10)$$

$$\frac{\partial Var(\pi_{net\ buyer})}{\partial k} = \frac{\partial E(y_{hay})^2 Var(p_{hay}) + E(p_{hay})^2 Var(y_{hay}) + 2 E(y_{hay}) E(p_{hay}) Cov(y_{hay}, p_{hay}) + Var(p_{hay}) Var(y_{hay}) + Cov(y_{hay}, p_{hay})^2}{\partial k} + \frac{\partial d_{hay}^2 Var(p_{hay})}{\partial k} + \frac{\partial Cov(y_{hay}, p_{hay}, d_{hay}, p_{hay})}{\partial k} \quad (A.11)$$

Hence, considering the absence of hay price feedbacks, the marginal effects of species diversity on income variance under increasing drought risks can be expressed for our farm types as:

⁴⁴ Note that with price feedbacks (i.e. if many farmers adapt higher levels of species diversity), net buyers would gain additionally in terms of expected income compared to net sellers:

$$\frac{\partial^2 E(\pi_{net\ seller})}{\partial k \partial v} = \frac{\partial^2 E(y_{hay})}{\partial k \partial v} E(p_{hay}) + \frac{\partial^2 E(p_{hay})}{\partial k \partial v} E(y_{hay}) + \frac{\partial^2 E(y_{hay})}{\partial k \partial v} \frac{\partial^2 E(p_{hay})}{\partial k \partial v}$$

$$\frac{\partial^2 E(\pi_{net\ buyer})}{\partial k \partial v} = \frac{\partial^2 E(y_{hay}) - d_{hay}}{\partial k \partial v} E(p_{hay}) + \frac{\partial^2 E(p_{hay})}{\partial k \partial v} (E(y_{hay}) - d_{hay}) + \frac{\partial^2 E(y_{hay}) - d_{hay}}{\partial k \partial v} \frac{\partial^2 E(p_{hay})}{\partial k \partial v}$$

Where $\frac{\partial^2 E(p_{hay})}{\partial k \partial v} E(y_{hay}) < 0$ and $\frac{\partial^2 E(p_{hay})}{\partial k \partial v} (E(y_{hay}) - d_{hay}) \geq 0$ when $E(y_{hay}) \leq d_{hay}$. Moreover, $\frac{\partial^2 E(p_{hay})}{\partial k \partial v} E(y_{hay}) < \frac{\partial^2 E(p_{hay})}{\partial k \partial v} (E(y_{hay}) - d_{hay})$ also when $E(y_{hay}) \leq d_{hay}$ as long as $d_{hay} > 0$.

$$\frac{\partial^2 \text{Var}(\pi_{\text{net seller}})}{\partial k \partial v} = \frac{\partial^2 E(y_{\text{hay}})^2 \text{Var}(p_{\text{hay}}) + E(p_{\text{hay}})^2 \text{Var}(y_{\text{hay}}) + 2 E(y_{\text{hay}}) E(p_{\text{hay}}) \text{Cov}(y_{\text{hay}}, p_{\text{hay}}) + \text{Var}(p_{\text{hay}}) \text{Var}(y_{\text{hay}}) + \text{Cov}(y_{\text{hay}}, p_{\text{hay}})^2}{\partial k \partial v} \quad (\text{A.12})$$

$$\frac{\partial^2 \text{Var}(\pi_{\text{net buyer}})}{\partial k \partial v} = \frac{\partial^2 E(y_{\text{hay}})^2 \text{Var}(p_{\text{hay}}) + E(p_{\text{hay}})^2 \text{Var}(y_{\text{hay}}) + 2 E(y_{\text{hay}}) E(p_{\text{hay}}) \text{Cov}(y_{\text{hay}}, p_{\text{hay}}) + \text{Var}(p_{\text{hay}}) \text{Var}(y_{\text{hay}}) + \text{Cov}(y_{\text{hay}}, p_{\text{hay}})^2}{\partial k \partial v} + \frac{\partial^2 \text{Cov}(y_{\text{hay}} p_{\text{hay}}, d_{\text{hay}} p_{\text{hay}})}{\partial k \partial v} \quad (\text{A.13})$$

The marginal effects of species diversity on income variance differ between the two farm types by $\frac{\partial^2 \text{Cov}(y_{\text{hay}} p_{\text{hay}}, d_{\text{hay}} p_{\text{hay}})}{\partial k \partial v}$, which is a negative term.⁴⁵ Thus, the marginal certainty equivalent of species diversity increases more with increasing drought risks for net buyers than for net sellers:

$$\frac{\partial^2 CE_{\text{net buyer}}}{\partial k \partial v} > \frac{\partial^2 CE_{\text{net seller}}}{\partial k \partial v} > 0 \quad (\text{A.14})$$

We can therefore prove that increasing drought risks increase the optimal species diversity level of net buyers more than that of net sellers, following equation A.5 to A.7:

$$\frac{\partial v_{\text{net buyer}}^*}{\partial k} > \frac{\partial v_{\text{net seller}}^*}{\partial k} > 0 \quad (\text{A.15})$$

4.12.3 References – Appendix 1

- Bohrnstedt, G. W. & Goldberger, A. S. (1969). On the exact covariance of products of random variables. *Journal of the American Statistical Association*, **64**, 1439-1442. <https://doi.org/10.1080/01621459.1969.10501069>
- Burt, O. R. & Finley, R. M. (1968). Statistical analysis of identities in random variables. *American Journal of Agricultural Economics*, **50**, 734-744. <https://doi.org/10.2307/1238271>

⁴⁵ Note that with price feedbacks net buyers would additionally gain in terms of expected income variance compared to net sellers given that:

$$\frac{\partial^2 d_{\text{hay}} \text{Var}(p_{\text{hay}})}{\partial k \partial v} < 0 \text{ when } d_{\text{hay}} > 0.$$

4.13 Appendix 2 – Empirical Framework

4.13.1 Drought Variable Definition

To identify droughts in our hay price time series, we use the Standardized Precipitation Evapotranspiration Index (SPEI), which contains precipitation and potential evapotranspiration information (Vicente-Serrano et al. 2010). We use a SPEI that comprises information about the last three months (3-month SPEI) and we consider only droughts in the entire summer period (i.e. July and August). We define SPEI values below the threshold of -1.5 as droughts. This is a threshold representing ‘severe droughts’ (Yu et al. 2014). Hence, we define our drought variable, γ , as:

$$\gamma = \begin{cases} \left| \frac{SPEI_{July} + SPEI_{August}}{2} \right|, & SPEI_{July} < -1.5 \text{ and } SPEI_{August} < -1.5 \\ 0, & \text{else} \end{cases} \quad (\text{B.1})$$

4.13.2 Implementation of Bivariate Normal Distribution in the Simulation

In the simulation we draw hay yields and prices using a bivariate normal distribution based on conditional normal distribution (Bertsekas and Tsitsiklis 2002):

$$\hat{y}_{hay,g,\xi} \sim N\left(E(y_{hay}), Var(y_{hay})\right) \quad (\text{B.2})$$

$$\hat{p}_{hay,g,\xi} \sim N\left(E(p_{hay}|y_{hay}), Var(\hat{p}_{hay})\right) \quad (\text{B.3})$$

where:

$$E(p_{hay}|y_{hay}) = E(p_{hay}) + \rho \left(Var(p_{hay})^{0.5} / Var(y_{hay})^{0.5}\right) (y_{hay} - E(y_{hay})) \quad (\text{B.4})$$

$$Var(\hat{p}_{hay}) = (1 - \rho^2) Var(p_{hay}) \quad (\text{B.5})$$

Note that g indicates one draw, i.e. year, of a 25-year period of the simulation, ξ . We assume a correlation coefficient, ρ , of -0.6 (Ramsey et al. 2019). We use this form of the bivariate distribution as it allows us, in combination with ‘set.seed’, to draw first hay yields for the average monoculture and corresponding hay prices and, second, to draw hay yields with different diversity level while considering the same correlations with hay prices as for the average monocultures.

4.13.3 Regression Analysis of Simulation Results

We use the following regression models for analyzing the simulated expected income, risk premium and certainty equivalent:

$$E(\hat{\pi}_\xi) = \beta_0 + \beta_k k_\xi + \beta_\theta \theta_\xi + \beta_{k\theta} k_\xi \theta_\xi + \beta_{k\theta v} k_\xi \theta_\xi v_\xi^{0.5} + e_{3\xi} \quad (\text{B.6})$$

$$\begin{aligned} \widehat{Risk\ premium}_\xi = & \beta_0 + \beta_k k_\xi + \beta_\Theta \Theta_\xi + \beta_r r_\xi + \beta_{k\Theta} k_\xi \Theta_\xi + \beta_{kr} k_\xi r_\xi + \beta_{\Theta r} \Theta_\xi r_\xi + \beta_{k\Theta r} k_\xi \Theta_\xi r_\xi + \\ & \beta_{k\Theta r v} k_\xi \Theta_\xi r_\xi v_\xi^{0.5} + e_{4\xi} \end{aligned} \quad (B.7)$$

$$\begin{aligned} \widehat{Certainty\ equivalent}_\xi = & \beta_0 + \beta_k k_\xi + \beta_\Theta \Theta_\xi + \beta_r r_\xi + \beta_{k\Theta} k_\xi \Theta_\xi + \beta_{kr} k_\xi r_\xi + \beta_{\Theta r} \Theta_\xi r_\xi + \\ & \beta_{k\Theta r} k_\xi \Theta_\xi r_\xi + \beta_{k\Theta r v} k_\xi \Theta_\xi r_\xi v_\xi^{0.5} + e_{5\xi} \end{aligned} \quad (B.8)$$

The index ξ refers to the 25-year period of the simulation. k_ξ refers to a specific scenario, Θ_ξ to the farm type, i.e. net position of farmers, r_ξ to the Arrow-Pratt risk coefficient of risk aversion, and $v_\xi^{0.5}$ to the square root of species diversity.

4.13.4 Calculation of Drought Effects

We followed equation 25 to calculate the drought effects, with $M = 12$, and $M_1 = 9$ (Hofer et al. 2016):

i) Effect on hay prices: *drought price effect*_{*m*} = 14.22% (this is the mean for all *m* in M_1), *annual drought price effect* = $(\sum_{m=1}^{M_1} \text{drought price effect}_m) / M = 10.67\%$.

ii) Effect on lower confidence interval (LCI) of hay prices: *drought LCI price effect*_{*m*} = 4.49% (this is the mean for all *m* in M_1), *annual drought LCI price effect* = $(\sum_{m=1}^{M_1} \text{drought LCI price effect}_m) / M = 3.37\%$.

iii) Effect on upper confidence interval (UCI) of hay prices: *drought UCI price effect*_{*m*} = 23.66% (this is the mean for all *m* in M_1), *annual drought UCI price effect* = $(\sum_{m=1}^{M_1} \text{drought UCI price effect}_m) / M = 17.74\%$.

4.13.5 References – Appendix 2

Bertsekas, D. P. & Tsitsiklis, J. N. (2002). *Introduction to Probability*, first ed. Athena Scientific, Belmont, USA.

Hofer, D. et al. (2016). Yield of temperate forage grassland species is either largely resistant or resilient to experimental summer drought. *Journal of Applied Ecology*, **53**, 1023-1034. <https://doi.org/10.1111/1365-2664.12694>

Ramsey, A. F., Goodwin, B. K. & Ghosh, S. K. (2019). How high the hedge: relationships between prices and yields in the federal crop insurance program. *Journal of Agricultural and Resource Economics*, **44**, 227-245. <https://doi.org/10.22004/ag.econ.287967>

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4.14 Appendix 3 – Figures and Tables

4.14.1 Figures

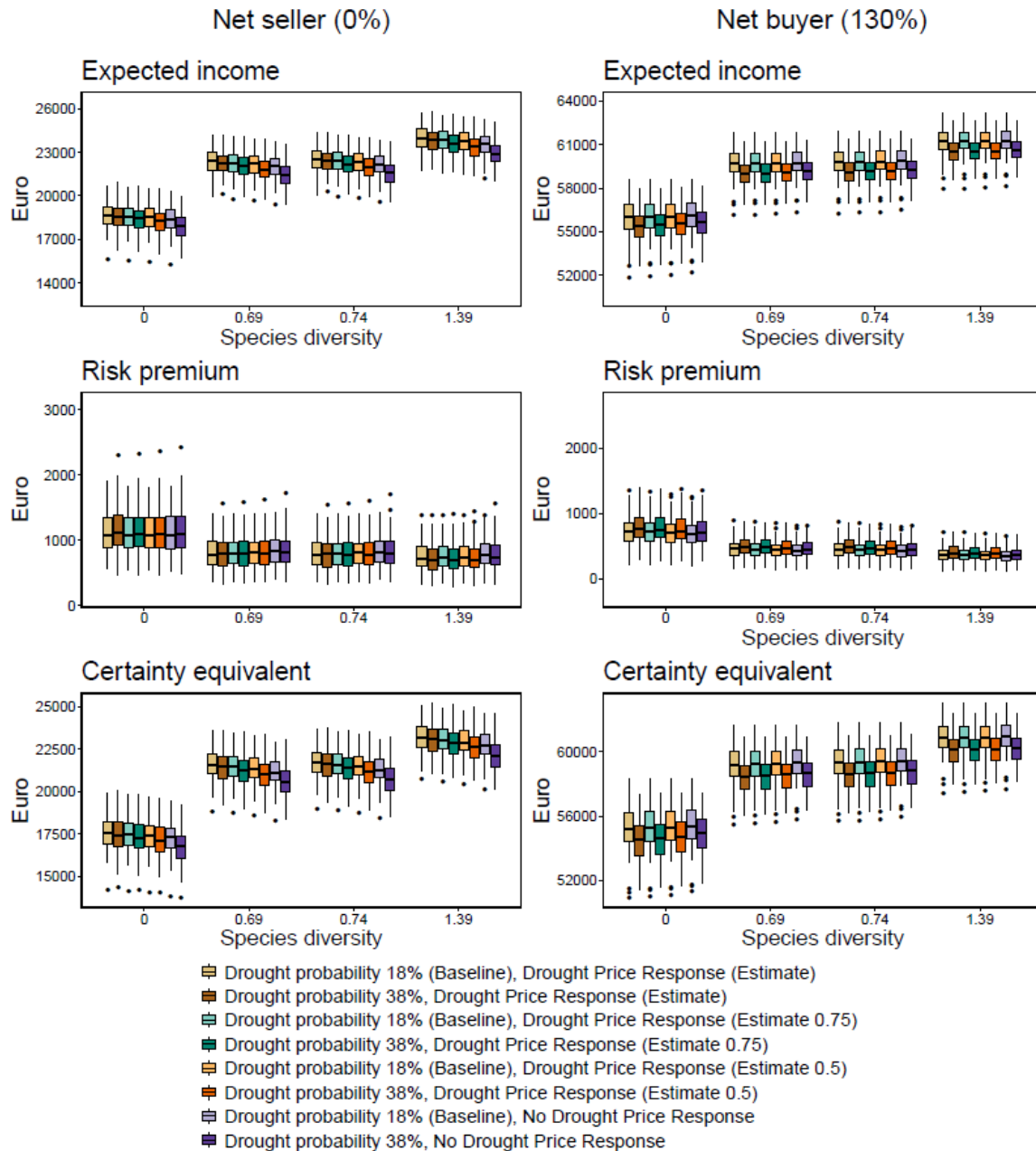


Figure A1: Effects of increasing drought risks (i.e. increasing drought probability) and species diversity on expected income, risk premium and certainty equivalent (in Euro farm⁻¹ year⁻¹) considering different drought price responses. We consider the estimated drought price response, fractions of the estimated drought price response (i.e. 75% and 50%) and no drought price response. The Arrow-Pratt risk coefficient of risk aversion, r , was assumed to be 2.

Note: We assume a farm size of 16.5 ha, which is based on the average farm size in South Germany (Table A1). In the drought probability 18% (baseline) scenario considering the average monoculture the mean net sellers' expected income (risk premium) was 18669 (1125) Euro farm⁻¹ year⁻¹. The

respective value for the average net buyers was 55940 (727) Euro farm⁻¹ year⁻¹. All farms are assumed to receive an additional income of 20000 Euro farm⁻¹. See Table A1 and A6 for calculation basis.

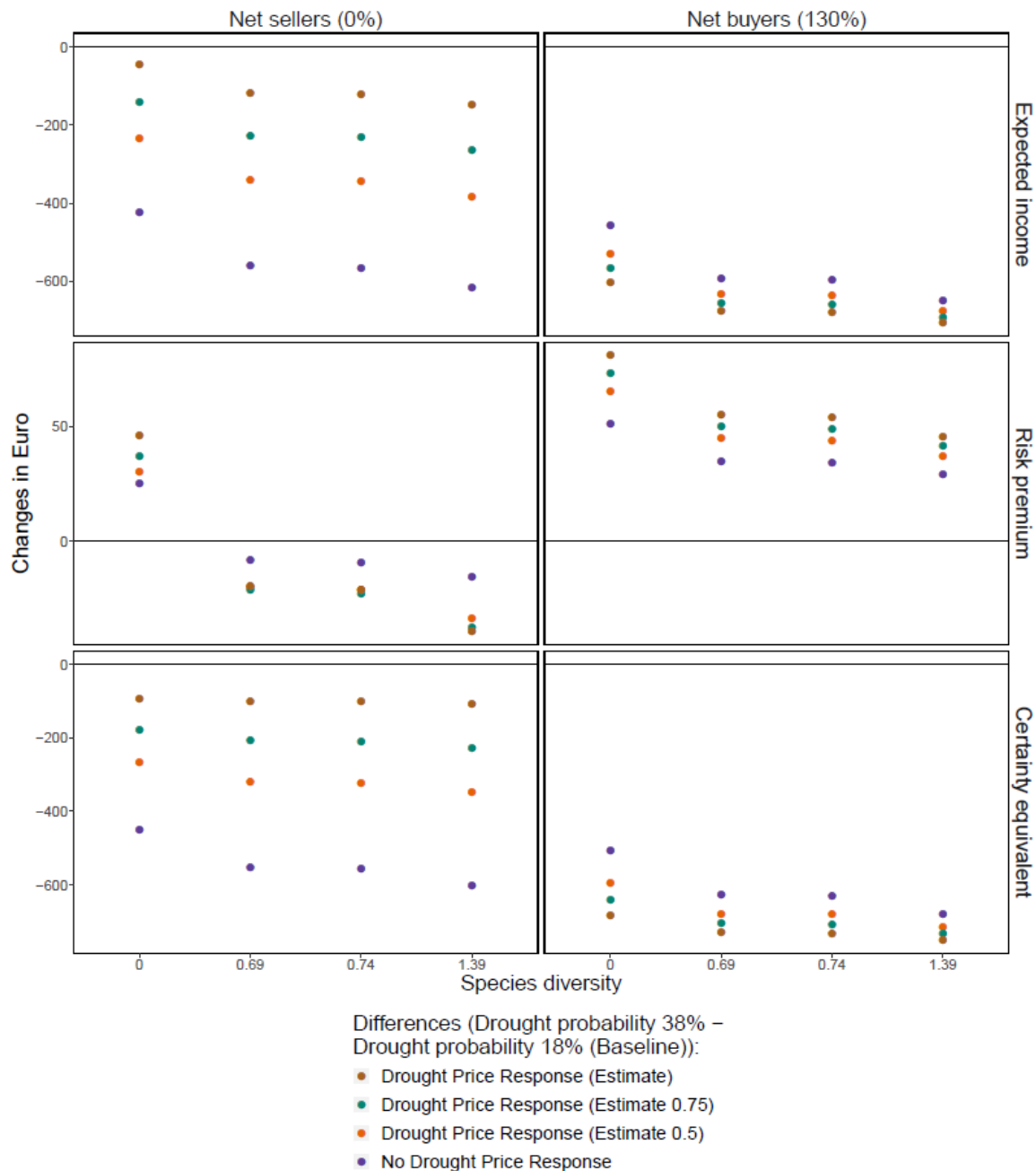


Figure A2: Average differences of the effects of increasing drought risks (i.e. increasing drought probability) and species diversity on expected income, risk premium and certainty equivalent (in Euro farm⁻¹ year⁻¹) between drought probability 38% and drought probability 18% (baseline) considering different drought price responses. We consider the estimated drought price response, fractions of the estimated drought price response (i.e. 75% and 50%) and no drought price response. The Arrow-Pratt risk coefficient of risk aversion, r , was assumed to be 2.

Note: We assume a farm size of 16.5 ha, which is based on the average farm size in South Germany (Table A1). In the drought probability 18% (baseline) scenario considering the average monoculture the mean net sellers' expected income (risk premium) was 18669 (1125) Euro farm⁻¹ year⁻¹. The respective value for the average net buyers was 55940 (727) Euro farm⁻¹ year⁻¹. All farms are assumed to receive an additional income of 20000 Euro farm⁻¹. See Table A1 and A6 for calculation basis.

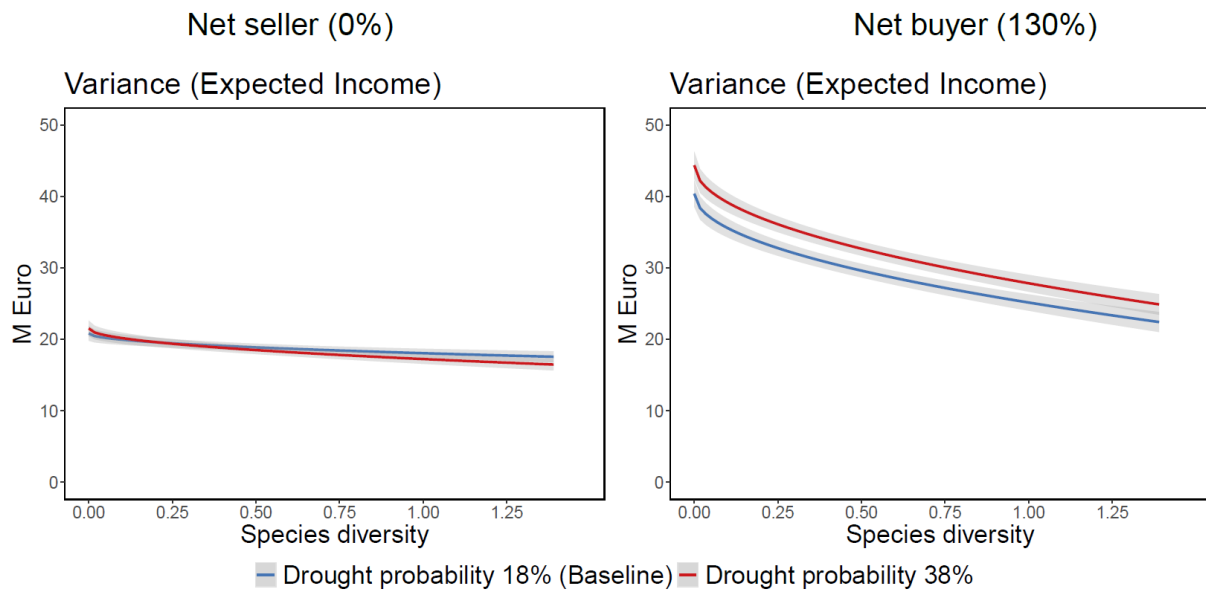


Figure A3: Drought and diversity effects on variance in Million Euro farm⁻¹ year⁻¹ over a range of risk aversion coefficients. *Note:* The average farm size is 16.5 ha and all farms are assumed to receive an additional income of 20000 Euro farm⁻¹ (Table A1).

4.14.2 Tables

Table A1: Simulation input - farm and market condition.

Variable	Unit	Value	Remark	Source
Milk prices	Euro 100 kg ⁻¹	36	Average values in South Germany in 2018	BMEL 2019
Cows	Number of cows	42	Average value in South Germany in 2018	Destatis 2019
Net milk production for sale	kg cow ⁻¹ year ⁻¹	7403	Average values in Bavaria based on Simmental cows (milk production – milk for feeding calves)	LfL 2019a
Hay demand (% of total feed)	100 kg cow ⁻¹ year ⁻¹	41.98 (62%)	We assume that only hay is used as roughage.	LfL 2019a
Total hay demand	100 kg year ⁻¹	Hay demand x cows		
Grassland	ha	16.54 = Total hay demand/mean hay yields	Total grassland area is based on total hay demand and expected hay yields of the drought probability 18% (baseline) scenario considering the average monoculture.	
Total hay yields	100 kg year ⁻¹	Hay yields x grassland		
Total milk production	100 kg year ⁻¹	Net milk production for sale x cows/100		
Net hay demand	100 kg year ⁻¹	Total hay yields - total hay demand		
Other variable milk production costs	Euro cow ⁻¹ year ⁻¹	1537.6	Average value in Bavaria	LfL 2019a
Variable costs for maintaining grasslands and for hay production	Euro ha ⁻¹ year ⁻¹	1155.8	Average value in Bavaria. We assume 5 cuts per year (as the number of cuts at the Switzerland 2 and Ireland site), a plot size of 5 ha, and that farmers convert all of their grass into hay.	LfL 2019b, 2019c
Additional fixed income	Euro year ⁻¹	20000	These payments can represent direct payments or other farm income.	
Arrow-Pratt risk coefficient of risk aversion	-U''/U'	Ranges from 0.5 to 4	The range covers low to very risk averse farmers.	Hardaker et al. 2015

Remark: All values are without value added tax.

Table A2: Summary statistics for experimental hay yields in 100 kg ha⁻¹ year⁻¹.

Shannon-index	Drought (1=true, 0=false)	Hay yields		
		Switzerland 1	Switzerland 2	Ireland
0	0	106.58 (30.99)	104.25 (18.99)	91.93 (21.71)
0	1	95.9 (29.02)	100.09 (26.42)	82.72 (18.74)
0.69	0	132.05 (26.46)	120.11 (19.68)	103.57 (24.73)
0.69	1	121.45 (21.19)	111.6 (21.28)	88.58 (23.87)
0.74	0	148.5 (14.1)	-	115.21 (24.87)
0.74	1	134.18 (15.05)	-	95.35 (26.4)
1.39	0	156.2 (9.69)	140.82 (22.33)	108.49 (21.36)
1.39	1	139.36 (14.57)	131.36 (14.15)	96.23 (24.17)

Note: Hay yield values are means. Standard deviations are given in parentheses.

Table A3: Augmented Dickey–Fuller unit root test.

Test statistic	τ_3	ϕ_2 (constant)
Hay prices	-3.08**	4.92**
Droughts	-8.06***	-

Note: *, **, *** denote significance at the 10%, 5% and 1% level, respectively. τ_3 is the test statistic for unit root. Null hypothesis is presence of a unit root. For critical values see Table A4.

Table A4: Critical values for the Augmented Dickey–Fuller unit root test.

	With a constant		Without a constant
	τ_3	ϕ_2 (constant)	τ_3
1%	-3.46	6.52	-2.58
5%	-2.88	4.63	-1.95
10%	-2.57	3.81	-1.62

Table A5: Regression result for the mean and the variance of hay yields in 100 kg ha⁻¹ year⁻¹ (equation 23 and 24).

	Mean(hay yields)	Var(hay yields)
Intercept	123.74 (3.65)***	735.68 (75.56)***
Drought	-9.01 (3.74)**	
Species diversity ^{0.5} x Control	26.11 (3.43)***	-261.59 (63.27)***
Species diversity ^{0.5} x Drought	20.65 (3.13)***	-305.57 (62.58)***
Year number		
2	-19.42 (2.39)***	-89.69 (59.42)
3	-43.56 (2.41)***	-159.55 (51.42)***
Site		
2	-19.61 (3.51)***	-141.28 (94.9)
3	-16.14 (2.39)***	-265.85 (59.42)***
N	416	416
Adjusted R ²	0.57	0.13

Note: Numbers in parentheses are robust standard errors. * p < 0.1; ** p < 0.05; *** p < 0.01. Drought indicates $\gamma_i = 1$ and control indicates $\varphi_i = 1$. The interaction of species diversity ($v_i^{0.5}$) with control or drought are the coefficient estimates of $\beta_{\varphi v}$ and $\beta_{\gamma v}$ (see equation 20 and 21).

Table A6: Hay yields and their variance as well as hay prices and their variance in the different scenarios.

		Drought probability					
		0%	18 % (Baseline)	28%	38%	48%	100%
Species diversity							
Drought probability		0%	18%	28%	38%	48%	100%
Expected hay yields (100 kg ha ⁻¹ year ⁻¹)	0	105.19	102.66	102.11	103.21	105.96	97.26
	0.69	126.05	122.66	121.68	123.69	127.72	114.46
	0.74	126.8	123.37	122.38	124.42	128.5	115.07
	1.39	134.8	131.04	129.89	132.29	136.85	121.67
Variance of hay yields (100 kg ha ⁻¹ year ⁻¹)	0	969.56	987.35	997.38	989.35	930.28	969.26
	0.69	761.51	772.11	787.77	777.15	719.8	713.09
	0.74	754.21	764.66	780.47	769.76	712.31	703.97
	1.39	676.04	685.35	702.5	690.87	631.54	605.67
Expected hay prices (Euro 100 kg ⁻¹)		10.6	10.84	10.97	10.81	10.44	11.5
Variance of hay prices (Euro 100 kg ⁻¹)		4.49	4.4	4.45	4.64	4.08	4.34

Table A7: Regression result for the expected income in Euro farm⁻¹ year⁻¹ (equation B.10).

	Expected income
Intercept	18667.93 (87.99)***
Scenario(k)	
Drought probability 0%	-116.64 (118.9)
Drought probability 28%	-3.81 (126.72)
Drought probability 38%	-46.78 (122.48)
Drought probability 48%	57.69 (121.01)
Drought probability 100%	60.61 (119.84)
Farm type(θ)	
Net buyer	37271.62 (145.16)***
Scenario(k) x Farm type(θ)	
Drought probability 0% x Net buyer	366.46 (195.89)*
Drought probability 28% x Net buyer	-490.05 (205.3)**
Drought probability 38% x Net buyer	-557.13 (205.15)***
Drought probability 48% x Net buyer	-850.74 (207.7)***
Drought probability 100% x Net buyer	-2078.95 (208.75)***
Scenario(k) x Farm type(θ) x Species diversity(v) ^{0.5}	
Drought probability 0% x Net seller x Species diversity ^{0.5}	4616.12 (93.58)***
Drought probability 18% (Baseline) x Net seller x Species diversity ^{0.5}	4483.4 (103.65)***
Drought probability 28% x Net seller x Species diversity ^{0.5}	4488.04 (105.08)***
Drought probability 38% x Net seller x Species diversity ^{0.5}	4396.98 (98.63)***
Drought probability 48% x Net seller x Species diversity ^{0.5}	4352.3 (95.09)***
Drought probability 100% x Net seller x Species diversity ^{0.5}	4056.2 (93.98)***
Drought probability 0% x Net buyer x Species diversity ^{0.5}	4616.12 (117.04)***
Drought probability 18% (Baseline) x Net buyer x Species diversity ^{0.5}	4483.4 (129.75)***
Drought probability 28% x Net buyer x Species diversity ^{0.5}	4488.04 (127.1)***
Drought probability 38% x Net buyer x Species diversity ^{0.5}	4396.98 (131.91)***
Drought probability 48% x Net buyer x Species diversity ^{0.5}	4352.3 (138.95)***
Drought probability 100% x Net buyer x Species diversity ^{0.5}	4056.2 (140.73)***
N	4800
Adjusted R ²	0.9974

Note: Numbers in parentheses are robust standard errors. * p < 0.1; ** p < 0.05; *** p < 0.01.

Table A8: Regression result for the risk premium and the certainty equivalent in Euro farm⁻¹ year⁻¹ (equation B.11 and B.12).

	Risk premium	Certainty equivalent
Intercept	278.4 (7.74)***	18389.53 (88.32)***
Scenario(k)		
Drought probability 0%	-27.47 (10.48)***	-89.17 (119.74)
Drought probability 28%	16.28 (11.26)	-20.09 (128)
Drought probability 38%	11.13 (11.67)	-57.92 (124.73)
Drought probability 48%	19.56 (11.87)*	38.14 (122.97)
Drought probability 100%	40.74 (12.57)***	19.88 (122.58)
Farm type(θ)		
Net buyer	-97.53 (9.57)***	37369.15 (147.56)***
Risk coefficient(r)		
r2	835.19 (31.92)***	-835.19 (128.44)***
r4	1948.77 (62.41)***	-1948.77 (139.17)***
Scenario(k) Farm type(θ)		
Drought probability 0% x Net buyer	11.93 (13.28)	354.53 (199.29)*
Drought probability 28% x Net buyer	1.86 (14.14)	-491.91 (209.65)**
Drought probability 38% x Net buyer	8.99 (14.28)	-566.12 (210.05)***
Drought probability 48% x Net buyer	8.89 (14.95)	-859.62 (212.14)***
Drought probability 100% x Net buyer	8.83 (15.95)	-2087.77 (214.25)***
Scenario(k) Risk coefficient(r)		
Drought probability 0% x r2	-82.41 (43.21)*	82.41 (174.92)
Drought probability 28% x r2	48.85 (46.41)	-48.85 (187.8)
Drought probability 38% x r2	33.4 (48.13)	-33.4 (185.52)
Drought probability 48% x r2	58.67 (48.96)	-58.67 (182.69)
Drought probability 100% x r2	122.22 (51.83)**	-122.22 (184.22)
Drought probability 0% x r4	-192.28 (84.49)**	192.28 (190.34)
Drought probability 28% x r4	113.98 (90.76)	-113.98 (205.31)
Drought probability 38% x r4	77.94 (94.11)	-77.94 (206.48)
Drought probability 48% x r4	136.89 (95.73)	-136.89 (203.72)
Drought probability 100% x r4	285.18 (101.34)***	-285.18 (208.66)
Farm type(θ) Risk coefficient(r)		
Net buyer x r2	-292.59 (39.46)***	292.59 (216.23)
Net buyer x r4	-682.72 (77.16)***	682.72 (231.51)***
Scenario(k) Farm type(θ) Risk coefficient(r)		
Drought probability 0% x Net buyer x r2	35.8 (54.77)	-35.8 (292.53)
Drought probability 28% x Net buyer x r2	5.57 (58.28)	-5.57 (309.38)
Drought probability 38% x Net buyer x r2	26.96 (58.87)	-26.96 (311.1)
Drought probability 48% x Net buyer x r2	26.66 (61.63)	-26.66 (313.51)
Drought probability 100% x Net buyer x r2	26.47 (65.77)	-26.47 (319.14)
Drought probability 0% x Net buyer x r4	83.54 (107.1)	-83.54 (314.18)
Drought probability 28% x Net buyer x r4	13 (113.96)	-13 (334.26)

	Drought probability 38% x Net buyer x r4	62.92 (115.12)	-62.92 (337.47)
	Drought probability 48% x Net buyer x r4	62.21 (120.5)	-62.21 (339.98)
	Drought probability 100% x Net buyer x r4	61.77 (128.6)	-61.77 (349.8)
Scenario(k)	Farm type(θ)	Risk coefficient(r)	Species diversity(v) ^{0.5}
	Drought probability 0% x Net seller x r0.5 x Species diversity ^{0.5}	-73.03 (7.72)***	4689.15 (94.39)***
	Drought probability 18% (Baseline) x Net seller x r0.5 x Species diversity ^{0.5}	-83.28 (8.48)***	4566.68 (103.95)***
	Drought probability 28% x Net seller x r0.5 x Species diversity ^{0.5}	-97.04 (8.9)***	4585.08 (106.24)***
	Drought probability 38% x Net seller x r0.5 x Species diversity ^{0.5}	-101.75 (9.56)***	4498.72 (101.49)***
	Drought probability 48% x Net seller x r0.5 x Species diversity ^{0.5}	-105.57 (9.67)***	4457.87 (97.4)***
	Drought probability 100% x Net seller x r0.5 x Species diversity ^{0.5}	-127.73 (10.64)***	4183.93 (97.3)***
	Drought probability 0% x Net buyer x r0.5 x Species diversity ^{0.5}	-73.19 (6.3)***	4689.3 (119.5)***
	Drought probability 18% (Baseline) x Net buyer x r0.5 x Species diversity ^{0.5}	-76.41 (6.03)***	4559.81 (132.66)***
	Drought probability 28% x Net buyer x r0.5 x Species diversity ^{0.5}	-83.27 (6.89)***	4571.31 (130.95)***
	Drought probability 38% x Net buyer x r0.5 x Species diversity ^{0.5}	-83.97 (6.41)***	4480.95 (135.57)***
	Drought probability 48% x Net buyer x r0.5 x Species diversity ^{0.5}	-85.06 (7.64)***	4437.36 (142.07)***
	Drought probability 100% x Net buyer x r0.5 x Species diversity ^{0.5}	-99.66 (8.58)***	4155.86 (144.96)***
	Drought probability 0% x Net seller x r2 x Species diversity ^{0.5}	-292.13 (30.87)***	4908.24 (100.41)***
	Drought probability 18% (Baseline) x Net seller x r2 x Species diversity ^{0.5}	-333.1 (33.92)***	4816.5 (108.87)***
	Drought probability 28% x Net seller x r2 x Species diversity ^{0.5}	-388.15 (35.61)***	4876.18 (113.91)***
	Drought probability 38% x Net seller x r2 x Species diversity ^{0.5}	-406.98 (38.23)***	4803.96 (114.52)***
	Drought probability 48% x Net seller x r2 x Species diversity ^{0.5}	-422.29 (38.68)***	4774.59 (109.26)***
	Drought probability 100% x Net seller x r2 x Species diversity ^{0.5}	-510.91 (42.55)***	4567.12 (112.81)***
	Drought probability 0% x Net buyer x r2 x Species diversity ^{0.5}	-292.75 (25.19)***	4908.86 (128.47)***
	Drought probability 18% (Baseline) x Net buyer x r2 x Species diversity ^{0.5}	-305.63 (24.14)***	4789.03 (142.56)***
	Drought probability 28% x Net buyer x r2 x Species diversity ^{0.5}	-333.09 (27.56)***	4821.12 (143.88)***
	Drought probability 38% x Net buyer x r2 x Species diversity ^{0.5}	-335.89 (25.65)***	4732.87 (147.69)***
	Drought probability 48% x Net buyer x r2 x Species diversity ^{0.5}	-340.23 (30.57)***	4692.52 (153.36)***
	Drought probability 100% x Net buyer x r2 x Species diversity ^{0.5}	-398.64 (34.31)***	4454.84 (159.75)***
	Drought probability 0% x Net seller x r4 x Species diversity ^{0.5}	-584.26 (61.73)***	5200.37 (115.38)***
	Drought probability 18% (Baseline) x Net seller x r4 x Species diversity ^{0.5}	-666.2 (67.84)***	5149.6 (123.55)***

Drought probability 28% x Net seller x r4 x Species diversity ^{0.5}	-776.29 (71.22)***	5264.33 (132.08)***
Drought probability 38% x Net seller x r4 x Species diversity ^{0.5}	-813.97 (76.46)***	5210.95 (139.37)***
Drought probability 48% x Net seller x r4 x Species diversity ^{0.5}	-844.57 (77.35)***	5196.87 (133.51)***
Drought probability 100% x Net seller x r4 x Species diversity ^{0.5}	-1021.83 (85.09)***	5078.03 (142.27)***
Drought probability 0% x Net buyer x r4 x Species diversity ^{0.5}	-585.5 (50.38)***	5201.61 (143.45)***
Drought probability 18% (Baseline) x Net buyer x r4 x Species diversity ^{0.5}	-611.27 (48.27)***	5094.67 (158.03)***
Drought probability 28% x Net buyer x r4 x Species diversity ^{0.5}	-666.17 (55.12)***	5154.21 (163.62)***
Drought probability 38% x Net buyer x r4 x Species diversity ^{0.5}	-671.78 (51.3)***	5068.76 (165.96)***
Drought probability 48% x Net buyer x r4 x Species diversity ^{0.5}	-680.45 (61.14)***	5032.75 (172.05)***
Drought probability 100% x Net buyer x r4 x Species diversity ^{0.5}	-797.28 (68.61)***	4853.48 (183.26)***
N	14400	14400
Adjusted R ²	0.788	0.997

Note: Numbers in parentheses are robust standard errors. * $p < 0.1$; ** $p < 0.05$; *** $p < 0.01$. Risk coefficient refers to the Arrow-Pratt risk coefficient. We indicate for readability in this table the Arrow-Pratt risk coefficient of value 0.5, 2 and 4 as r0.5, r2 and r4, respectively.

Table A9: Effect of increasing drought risks and additional diversity effects under increasing drought risks on the expected income.

Relation-ship	Scenario (k)	Farm type (θ)	Expected income
Effect of increasing drought risks	Drought probability 0%	Net seller	-116.64 (0.962)
	Drought probability 28%	Net seller	-3.81 (0.001)
	Drought probability 38%	Net seller	-46.78 (0.146)
	Drought probability 48%	Net seller	57.69 (0.227)
	Drought probability 100%	Net seller	60.61 (0.256)
	Drought probability 0%	Net buyer	249.82 (2.575)
	Drought probability 28%	Net buyer	-493.85 (9.349)***
	Drought probability 38%	Net buyer	-603.91 (13.466)***
	Drought probability 48%	Net buyer	-793.05 (22.071)***
	Drought probability 100%	Net buyer	-2018.33 (139.445)***
Additional diversity effects under increasing drought risks	Drought probability 0%	Net seller	132.72 (0.903)
	Drought probability 28%	Net seller	4.64 (0.001)
	Drought probability 38%	Net seller	-86.42 (0.365)
	Drought probability 48%	Net seller	-131.1 (0.869)
	Drought probability 100%	Net seller	-427.2 (9.322)***
	Drought probability 0%	Net buyer	132.72 (0.577)
	Drought probability 28%	Net buyer	4.64 (0.001)
	Drought probability 38%	Net buyer	-86.42 (0.218)
	Drought probability 48%	Net buyer	-131.1 (0.476)
	Drought probability 100%	Net buyer	-427.2 (4.981)**
	Drought probability 0%	Difference	0 (0)
	Drought probability 28%	Difference	0 (0)
	Drought probability 38%	Difference	0 (0)
	Drought probability 48%	Difference	0 (0)
	Drought probability 100%	Difference	0 (0)

Note: The estimates are the cumulative effects of the regression results (Table A7). Numbers in parentheses are χ^2 . χ^2 are based on a Wald-test-based comparison (Fox and Weisberg 2019). * $p < 0.1$; ** $p < 0.05$; *** $p < 0.01$. k is always compared to the drought probability 18% (baseline) scenario.

Table A10: Effect of increasing drought risks and additional diversity effects under increasing drought risks on the risk premium and the certainty equivalent.

Relation-ship	Scenario (k)	Farm type (θ)	Risk coefficient (r)	Risk premium	Certainty equivalent
Effect of increasing drought risks	Drought probability 0%	Net seller	2	-109.88 (6.871)***	-6.76 (0.003)
	Drought probability 28%	Net seller	2	65.13 (2.092)	-68.94 (0.252)
	Drought probability 38%	Net seller	2	44.54 (0.91)	-91.32 (0.442)
	Drought probability 48%	Net seller	2	78.22 (2.712)*	-20.53 (0.023)
	Drought probability 100%	Net seller	2	162.96 (10.504)***	-102.34 (0.554)
	Drought probability 0%	Net seller	0.5	-27.47 (6.871)***	-89.17 (0.555)
	Drought probability 28%	Net seller	0.5	16.28 (2.092)	-20.09 (0.025)
	Drought probability 38%	Net seller	0.5	11.13 (0.91)	-57.91 (0.216)
	Drought probability 48%	Net seller	0.5	19.56 (2.712)*	38.14 (0.096)
	Drought probability 100%	Net seller	0.5	40.74 (10.504)***	19.87 (0.026)
	Drought probability 0%	Net seller	4	-219.75 (6.871)***	103.11 (0.486)
	Drought probability 28%	Net seller	4	130.26 (2.092)	-134.07 (0.697)
	Drought probability 38%	Net seller	4	89.08 (0.91)	-135.86 (0.682)
	Drought probability 48%	Net seller	4	156.44 (2.712)*	-98.75 (0.37)
	Drought probability 100%	Net seller	4	325.92 (10.504)***	-265.3 (2.468)
	Drought probability 0%	Net buyer	2	-62.14 (3.621)*	311.96 (3.288)*
	Drought probability 28%	Net buyer	2	72.56 (4.503)**	-566.41 (9.758)***
	Drought probability 38%	Net buyer	2	80.49 (5.988)**	-684.4 (13.855)***
	Drought probability 48%	Net buyer	2	113.77 (9.817)***	-906.81 (23.474)***
	Drought probability 100%	Net buyer	2	198.26 (25.478)***	-2216.59 (132.67)***
	Drought probability 0%	Net buyer	0.5	-15.53 (3.621)*	265.36 (2.775)*
	Drought probability 28%	Net buyer	0.5	18.14 (4.503)**	-511.99 (9.509)***
	Drought probability 38%	Net buyer	0.5	20.12 (5.988)**	-624.03 (13.634)***
	Drought probability 48%	Net buyer	0.5	28.44 (9.817)***	-821.49 (22.584)***
	Drought probability 100%	Net buyer	0.5	49.56 (25.478)***	-2067.9 (138.481)***
	Drought probability 0%	Net buyer	4	-124.28 (3.621)*	374.1 (3.772)*
	Drought probability 28%	Net buyer	4	145.12 (4.503)**	-638.97 (9.719)***
	Drought probability 38%	Net buyer	4	160.98 (5.988)**	-764.89 (13.704)***
	Drought probability 48%	Net buyer	4	227.54 (9.817)***	-1020.58 (23.563)***
	Drought probability 100%	Net buyer	4	396.51 (25.478)***	-2414.85 (121.64)***
Additional diversity effects under increasing drought risks	Drought probability 0%	Net seller	2	40.97 (0.798)	91.74 (0.384)
	Drought probability 28%	Net seller	2	-55.04 (1.253)	59.68 (0.143)
	Drought probability 38%	Net seller	2	-73.88 (2.09)	-12.54 (0.006)
	Drought probability 48%	Net seller	2	-89.19 (3.006)*	-41.92 (0.074)
	Drought probability 100%	Net seller	2	-177.81 (10.678)***	-249.39 (2.53)
	Drought probability 0%	Net seller	0.5	10.24 (0.798)	122.47 (0.761)
	Drought probability 28%	Net seller	0.5	-13.76 (1.253)	18.4 (0.015)
	Drought probability 38%	Net seller	0.5	-18.47 (2.09)	-67.95 (0.219)
	Drought probability 48%	Net seller	0.5	-22.3 (3.006)*	-108.81 (0.583)
	Drought probability 100%	Net seller	0.5	-44.45 (10.678)***	-382.74 (7.226)***
	Drought probability 0%	Net seller	4	81.95 (0.798)	50.77 (0.09)
	Drought probability 28%	Net seller	4	-110.09 (1.253)	114.73 (0.402)

Drought probability 38%	Net seller	4	-147.77 (2.09)	61.34 (0.108)
Drought probability 48%	Net seller	4	-178.37 (3.006)*	47.27 (0.068)
Drought probability 100%	Net seller	4	-355.62 (10.678)***	-71.57 (0.144)
Drought probability 0%	Net buyer	2	12.89 (0.136)	119.83 (0.39)
Drought probability 28%	Net buyer	2	-27.45 (0.561)	32.09 (0.025)
Drought probability 38%	Net buyer	2	-30.26 (0.738)	-56.17 (0.075)
Drought probability 48%	Net buyer	2	-34.59 (0.789)	-96.51 (0.212)
Drought probability 100%	Net buyer	2	-93.01 (4.916)**	-334.19 (2.436)
Drought probability 0%	Net buyer	0.5	3.22 (0.136)	129.49 (0.526)
Drought probability 28%	Net buyer	0.5	-6.86 (0.561)	11.5 (0.004)
Drought probability 38%	Net buyer	0.5	-7.56 (0.738)	-78.86 (0.173)
Drought probability 48%	Net buyer	0.5	-8.65 (0.789)	-122.45 (0.397)
Drought probability 100%	Net buyer	0.5	-23.25 (4.916)**	-403.95 (4.226)**
Drought probability 0%	Net buyer	4	25.77 (0.136)	106.94 (0.251)
Drought probability 28%	Net buyer	4	-54.9 (0.561)	59.54 (0.069)
Drought probability 38%	Net buyer	4	-60.52 (0.738)	-25.91 (0.013)
Drought probability 48%	Net buyer	4	-69.18 (0.789)	-61.92 (0.07)
Drought probability 100%	Net buyer	4	-186.01 (4.916)**	-241.19 (0.993)
Drought probability 0%	Difference	2	-28.09 (0.238)	28.09 (0.013)
Drought probability 28%	Difference	2	27.59 (0.202)	-27.59 (0.012)
Drought probability 38%	Difference	2	43.62 (0.494)	-43.62 (0.028)
Drought probability 48%	Difference	2	54.59 (0.716)	-54.59 (0.044)
Drought probability 100%	Difference	2	84.81 (1.524)	-84.81 (0.102)
Drought probability 0%	Difference	0.5	-7.02 (0.238)	7.02 (0.001)
Drought probability 28%	Difference	0.5	6.9 (0.202)	-6.9 (0.001)
Drought probability 38%	Difference	0.5	10.91 (0.494)	-10.91 (0.002)
Drought probability 48%	Difference	0.5	13.65 (0.716)	-13.65 (0.003)
Drought probability 100%	Difference	0.5	21.2 (1.524)	-21.2 (0.008)
Drought probability 0%	Difference	4	-56.18 (0.238)	56.18 (0.043)
Drought probability 28%	Difference	4	55.18 (0.202)	-55.18 (0.036)
Drought probability 38%	Difference	4	87.25 (0.494)	-87.25 (0.087)
Drought probability 48%	Difference	4	109.19 (0.716)	-109.19 (0.136)
Drought probability 100%	Difference	4	169.61 (1.524)	-169.61 (0.306)

Note: The estimates are the cumulative effects of the regression results (Table A8). Numbers in parentheses are χ^2 . χ^2 are based on a Wald-test-based comparison (Fox and Weisberg 2019). * $p < 0.1$; ** $p < 0.05$; *** $p < 0.01$. k is always compared to the drought probability 18% (baseline) scenario. Risk coefficient refers to the Arrow-Pratt risk coefficient.

Table A11: Regression result of single year hay revenues in Euro farm⁻¹ year⁻¹.

Variable	Revenue
Intercept	17644.8 (81.778)***
Drought	271.3 (0.791)
Net buyer	-23539.9 (-57.995)***
Drought x Net buyer	-2772.3 (-4.342)***
Species diversity ^{0.5} x Drought x Net Seller	4602.7 (17.899)***
Species diversity ^{0.5} x Drought x Net Seller	4048.3 (12.834)***
Species diversity ^{0.5} x Drought x Net buyer	4279.2 (11.086)***
Species diversity ^{0.5} x Drought x Net buyer	4117.3 (8.921)***
N	4774
Adjusted R ²	0.86

Note: Numbers in parentheses are robust standard errors. * $p < 0.1$; ** $p < 0.05$; *** $p < 0.01$. The χ^2 of the net buyers' drought effects is 21.57 (Wald-test-based comparison). The revenues are without costs and revenues from milk sales.

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5 Chapter 5 – Drought Effects on Hay and Feed Grain Prices⁴⁶

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Abstract

Droughts represent a severe and increasing risk for the livestock sector as they can reduce hay and feed grain yields. Droughts are predicted to increase in frequency and magnitude under climate change. We here estimate the so far unexplored effect of drought shocks on feed prices. We use an empirical example from Germany and focus on the prices of hay as well as feed wheat and barley. Our results show that regional and national droughts substantially increase hay prices of up to 15%, start with a delay of about three months and last for about a year. In contrast, feed grain prices in our sample are not affected by regional or national droughts. These price responses can be linked to market integration, as the hay market are usually regionally organized while feed grains are traded transnationally. This knowledge is important to include into farm management and policy actions, especially considering climate change.

Keywords: hay prices, feed grain prices, droughts, weather extremes, market integration

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5.1 Introduction

Agriculture is highly vulnerable to droughts. This also holds for livestock production. Droughts can cause substantial reductions in yields of grassland and feed crops (e.g. Ciaï et al. 2005; Smit et al. 2008; Webber et al. 2018). Yet, the implications for the feed markets are not well studied, even if, under climate change such droughts are predicted to increase in frequency and magnitude (Dai 2013; IPCC 2013; Spinoni et al. 2018).

We estimate effects of droughts occurring on regional and national levels on feed prices using an empirical example from South Germany and focusing on important feed prices, including hay and feed wheat and barley prices. These prices are expected to be affected differently by shocks, considering differences in transport and transaction costs, thus potential market integration. Transport costs are here defined as costs occurring due to transport, e.g. for fuel and loading. Transaction costs include other costs that occur due to the exchange of goods, e.g. finding sellers or buyers and verification of quality. Hay, as an important feed source for dairy and beef sector as well as for feeding horses (Vanselow et al. 2012; LfL 2018), is a bulky commodity with varying quality, has a low per ton protein unit, is usually not transported over great distances and relatively low quantities are traded (Rudstrom 2004; McCullock et al. 2014). Thus, hay markets are rather regional, with relatively low transparency and a lack of formal market exchanges.⁴⁷ In contrast, feed wheat and barley, which are the two most important feed grains in Germany (BLE 2019), have typically higher protein unit per ton than hay, are transported over longer distances, larger quantities are traded and trade occurs transnationally (Liefert et al. 2010; Taheripour et al. 2011; BLE 2019). Thus, the feed grain market is over-regionally organized and is assumingly more transparent than the hay market. Depending on the animal, wheat and barley can be good substitutes for each other whereas hay is only limited a substitute for them given animals feed roughage and grain/concentrate ration requirements (Flanders and Gillespie 2015).

While previous studies looked at general hay prices dynamics (e.g. Bazen et al. 2008; McCullock et al. 2014; Peake et al. 2019), no study investigated the drought effects on hay prices. For major grain prices some studies explored the reaction to drought (e.g. Sternberg 2012; Chung et al. 2014). Other studies showed that grain prices positively react to anomalies in the El Niño-Southern Oscillation, which are linked to extreme weather events such as droughts (e.g. Algieri 2014; Ubilava 2017).

We contribute filling gaps in the literature by providing the first study on feed price dynamics, of different feed crops, in response to regional and national droughts. Our findings are important for private actors, such as farmers and insurances, as well as for public entities to improve management

⁴⁷ Note that in this paper market transparency refers to the availability, accuracy, timeliness and reliability of market information and formal market exchanges to the institutionalization and regularization of market exchanges.

of adverse drought effects. We found that droughts substantially increased hay prices while feed grain prices were not affected. These price responses can be linked to market integration.

In the remainder of the paper, we present our theoretical framework (1), which is followed by the description of the econometric framework (2) and the data (3). Next, we present our results of the baseline drought specification as well as of the robustness checks (4), and finally, we discuss and conclude our results (5).

5.2 Theoretical Framework

The demand and supply function for feed crops Q_{Dt} and Q_{St} , are summarized as follows (see e.g. Alam and Gilbert 2017):

$$Q_{D,t} = Q_D(P_t, H_t, V_t, \gamma_{1,t,r}) \quad (1)$$

$$Q_{S,t} = Q_S(P_t, H_t, V_t, \gamma_{2,t,r}) \quad (2)$$

P_t represents prices, for example wholesale prices, of the agricultural product, i.e. $P_t = [p_t^{wheat}, p_t^{barley}, p_t^{hay}]$, H_t is transport costs and V_t is transaction costs. Whether buyers or sellers bear the transport and transaction costs depends on the market (power) of the different parties (e.g. Graubner et al. 2011), therefore, we stated them explicitly in equation (1) and (2). $\gamma_{1,t,r}$ and $\gamma_{2,t,r}$ are vectors of variables: $\gamma_{1,t,r} = [Z_{1t}, k_{t,r}, \epsilon_{1,t}]$ and $\gamma_{2,t,r} = [Z_{2t}, k_{t,r}, \epsilon_{2,t}]$, where $Z_{1,t}$ and $Z_{2,t}$ are the respective demand and supply shifting variables. Note that we denote separately from the other demand and supply shifting variables droughts as $k_{t,r}$. We consider droughts at the regional level (i.e. in South Germany) or at the national level (i.e. in whole Germany)⁴⁸, i.e. $r = 1$ and $r = 2$ respectively. $\epsilon_{1,t}$ and $\epsilon_{2,t}$ are random shock variables.

Using equation (1) and (2) the change in storage, δ_t , can be expressed as:

$$\delta_t = Q_S(P_t, H_t, V_t, \gamma_{2,t,r}) - Q_D(P_t, H_t, V_t, \gamma_{1,t,r}) \quad (3)$$

Note that while we assume intra-annual adjustments of these storage levels, we expect no changes in storage levels across periods. Moreover, storage can be seen as part of the market characteristics and the presence of storage tends to buffer price shocks (Serra and Gil 2012).

We focus here on the impact of droughts on prices. Thus, using equation (3) we can obtain the inverse demand function, i.e. price function (sensu Alam and Gilbert 2017):

⁴⁸ We selected this resolution, because on the one hand we are interested in distinguishing drought effects on a smaller, i.e. regional, and larger, i.e. national, scale and on the other we consider the tendency of intra-national trade vis-à-vis cross-border trade of feed (McCallum 1995, Ghazalian 2012).

$$P_t = f(k_{t,r}, H_t, V_t, \bar{y}_{1,t}, \bar{y}_{2,t}, \delta_t) \quad (4)$$

where $\bar{y}_{1,t} = [Z_{1t}, \epsilon_{1,t}]$ and $\bar{y}_{2,t} = [Z_{2t}, \epsilon_{2,t}]$.

How prices in one region react to (drought) shocks, depend amongst others on costs for transport and transactions, as these costs affect market integration (Goodwin and Piggott 2001; Balcombe et al. 2007), thus, how production and price shocks in one region can be balanced by other regions. Costs for transport and transaction depend on distance between buyer and seller, Δs (for transaction costs because closer markets are usually better known), and are affected by droughts since droughts are systemic to a region. Additionally, transaction costs depend on the transparency of the market, ω . Furthermore, prices might not respond immediately but temporal delayed to shocks. The response time of a market to a shock, l_t , is assumed to depend on ω as well as on change in storage, δ_t . Hence, we can express the price function as:

$$P_t = f(k_{t,r}(H_t, V_t), H_t(\Delta s, k_{t,r}), V_t(\Delta s, k_{t,r}, \omega), l_t(\omega_t, \delta_t), \bar{y}_{1,t}, \bar{y}_{2,t}, \delta_t) \quad (5)$$

5.3 Economic Framework

To analyse the effect of droughts on the feed prices we use a structural vector autoregressive model (SVAR; see e.g. Lütkepohl 2005). SVAR models can be used to model the effect of an exogenous drought shock on endogenous feed prices using time series data.⁴⁹ Using a SVAR model allows identifying immediate and lagged drought effects on feed prices, therefore, we allow that market participants can adjust their prices expectation based on expected yields, thus also expected drought induced yield losses.⁵⁰ The SVAR is defined as:

$$AX_t = A_1^* X_{t-1} + \dots + A_d^* X_{t-d} + B \varepsilon_t \quad (6)$$

X_t is the vector of n variables in period t including a drought variable and feed prices, i.e. $X_t = [k_{t,r}, p_t^{wheat}, p_t^{barley}, p_t^{hay}]$, and d is the number of lags. A_j^* for $j = 1, \dots, d$ are the coefficient matrices ($n \times n$). B is an identity matrix, I_n , and ε_t is the structural error, which is assumed to be white noise. Multiplying equation (6) by the inverse of A results:

$$X_t = A^{-1} A_1^* X_{t-1} + \dots + A^{-1} A_d^* X_{t-d} + A^{-1} B \varepsilon_t \quad (7)$$

where $u_t = A^{-1} B \varepsilon_t$ is the vector of reduced form residuals and $\sum_u A^{-1} B B' A^{-1}$ its variance-covariance matrix. We restrict the model by using the 'canonical form' (see Appendix 1 for details).

⁴⁹ Previously, SVAR models were for example used to model effect of El Niño-Southern Oscillation or policy shocks (Alam and Gilbert 2017; Bastianin et al. 2018).

⁵⁰ Note that we assume that price expectations are connected to current prices, as they shift the demand curve to the right.

To identify the optimal length, d^* , we employ the Akaike information criterion (AIC). Furthermore, we used an Augmented Dickey–Fuller (ADF) unit root test with a constant to test for stationarity of the different price time series and without a constant to test for stationarity of the different drought time series (see e.g. Pfaff et al. 2016). Based on the estimated coefficients, we use impulse response functions to analyze the effect of drought shocks, i.e. ‘drought effects’, on prices. The impulse response functions show the effect over time of an exogenous impulse, here drought shock, on endogenous variables, here feed prices. They are useful as estimated SVAR coefficients alone are difficult to interpret. The shock to the impulse response function equals one standard deviation of the drought variable.⁵¹ This empirical framework allows identifying the different responses proposed in the theoretical framework, i.e. with respect to magnitude and timing of the response. Furthermore, the theoretical framework provides reason why prices react differently to droughts. Our analysis is conducted in R (R Core Team 2018) using the R-packages ‘vars’ and ‘urca’ (Pfaff 2008, Pfaff et al. 2016).⁵²

5.4 Data

5.4.1 Price Data

The price data contains prices of hay, feed wheat and barley from August 2002 to April 2019 from the German states of Bavaria and Baden-Württemberg, together referred to as ‘South Germany’ and was provided by the Bavarian Association of Farmers. South Germany represents about 30% of Germany’s hay production and 20% of its wheat and barley production⁵³ (Destatis 2019). Hay prices (Euro 100kg⁻¹) were reported as a bi-weekly average wholesale price ex-farm including value added tax for high-pressure pressed hay.⁵⁴ Feed wheat and barley prices (Euro 100kg⁻¹) were reported as weekly average wholesale purchasing prices from producers excluding value added tax. We converted prices into monthly natural long transformed real prices using the harmonized⁵⁵ index of consumer prices for Germany with base year 2015 (Eurostat 2019; Fig. 1, see Table A1 for summary statistics). These prices are henceforth indicated as hay, feed wheat and feed barley prices. The optimal lag length, d^* , of the price time series is 3 months based on the AIC and the ADF unit root test indicates that all price time series are stationary (Table A2).

⁵¹ We can obtain the coefficients of the impulse response functions from the following matrices (Lütkepohl 2005):
 $\Theta_j = \phi_j A^{-1} B, j = 1, \dots, d$

⁵² We used for the SVAR estimation the BFGS algorithm.

⁵³ Including all wheat and barley.

⁵⁴ Note that in Germany it is common that intensive grasslands are harvested four to five times per year (Socher et al. 2013).

⁵⁵ ‘Harmonized’ indicates that the index of consumer prices follows an EU-wide methodology (see e.g. Eurostat 2019 for definitions).

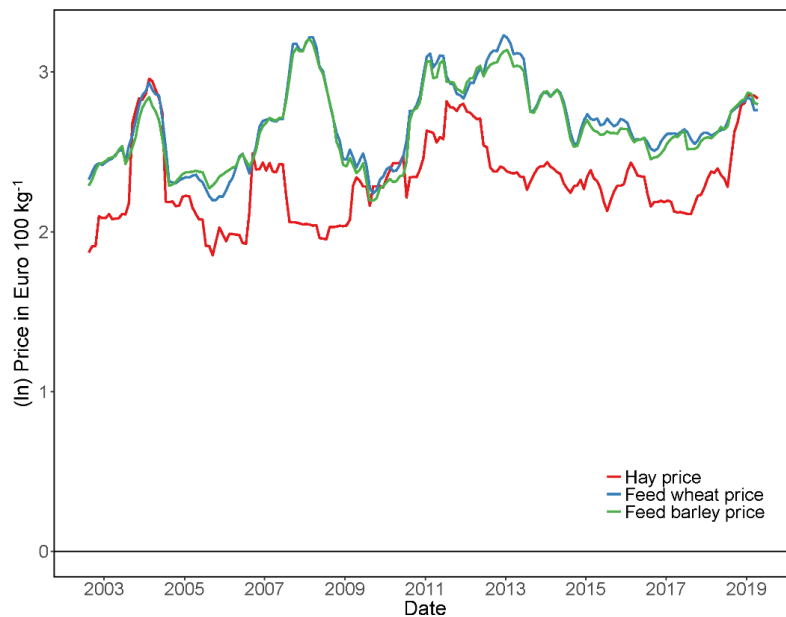


Figure 1: Prices of hay, feed wheat and feed barley.

5.4.2 Drought Information

For identifying droughts, we used the Standardized Precipitation Evapotranspiration Index (SPEI) as a standardized drought index. The SPEI incorporates information about precipitation and potential evapotranspiration (Vicente-Serrano et al. 2010). Thus, the SPEI also accounts for the impact of high temperature on drought intensity as temperature strongly affects evapotranspiration (Vicente-Serrano et al. 2010; Beguería et al. 2014). We used different SPEI lengths that comprise information about the last X months (SPEI- X). The drought variable were defined as drought, i.e. as $k_t = |\text{SPEI-}X|$, when the SPEI- X was below a specific threshold and otherwise as $k_t = 0$.

We focus on drought occurrence during the entire main vegetation period⁵⁶ (April – October). In the robustness checks, we also separately considered droughts in spring (April-May) and summer (June – August).⁵⁷

We used monthly potential evapotranspiration and precipitation data from January 1991 to April 2019 provided by German Meteorological Office as 1km x 1km gridded data (DWD 2019). The SPEI- X ⁵⁸ was calculated for every 1km x 1km grid of the agricultural area in i) South Germany and ii) whole Germany. For identifying the agricultural area⁵⁹ we used the 2012 ‘CORINE Land Cover 10 ha’ data (BKG 2019).

⁵⁶ In fact, while wheat and barley are usually winter crops, i.e. are planted in autumn, rainfall levels in autumn and winter are not limiting factors for yields (see e.g. Dalhaus et al. 2018).

⁵⁷ Droughts can cause at different times of the vegetation period losses for grain and hay yields (see e.g. Daryanto et al. 2017; Wilcox et al. 2017). The robustness checks also account for grains being more vulnerable to droughts in spring and grasslands in summer (see e.g. Denton et al. 2017; Dalhaus et al. 2018).

⁵⁸ For calculating the SPEI we used the R-packages ‘SPEI’ (Beguería and Vicente-Serrano 2017).

⁵⁹ The agricultural area considered includes the categories ‘non-irrigated arable land’, ‘pasture, meadows and other permanent grasslands under agricultural use’, ‘complex cultivation patterns’, ‘land principally occupied by

For both regions, South Germany and whole Germany, we calculated then the monthly average SPEI-X over all grid cells and the drought variable. The spatial aggregation of droughts are in line with its systemic nature, i.e. droughts usually affect larger areas (Miranda and Glauber 1997), and that market prices are an expression of the aggregated market supply and demand. All drought time series are stationary (Table A2).

The drought specification mainly used here reflects a ‘severe drought’, i.e. threshold = -1.5 (Yu et al. 2014), based on the SPEI-3. Fig. 2 shows severe droughts for South Germany and whole Germany for the different drought periods using SPEI-3. For this specification, the correlation between South Germany and whole Germany of the SPEI and severe droughts were 0.90 and 0.84, respectively (see Fig. A1 for more details). See Table 1 for additional specifications.

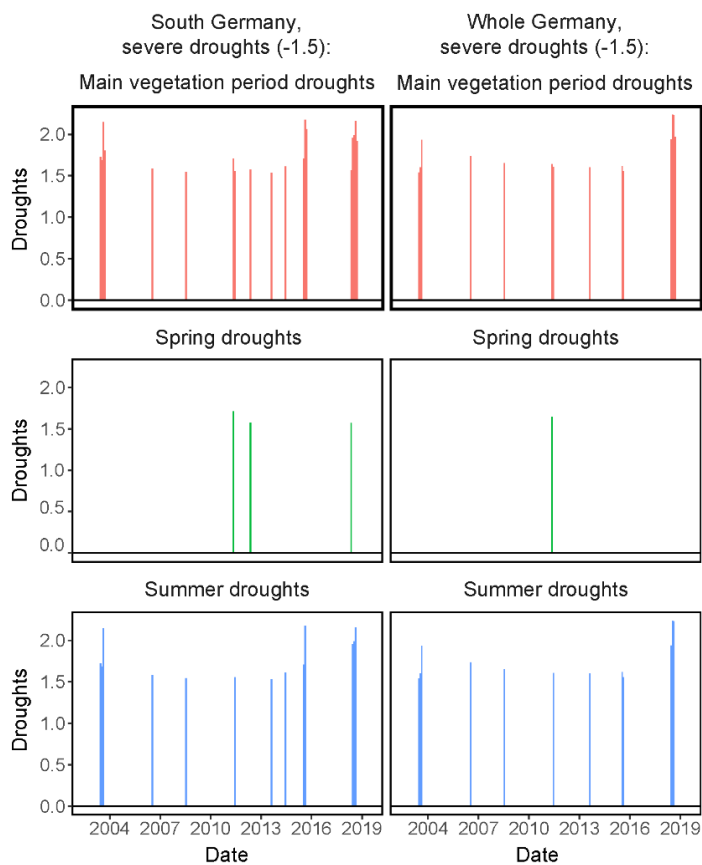


Figure 2: Severe droughts (threshold = -1.5) in South Germany and whole Germany for SPEI-3 and different drought periods. The bold frame indicates the baseline drought specification. See Fig. A2 to A7 for other drought specifications.

agriculture, with significant areas of natural vegetation’ and ‘natural grassland’. Note that we consider natural grasslands as they can be extensively grazed (Kosztra et al. 2019).

Table 1: Variation in drought specification.

Region	Drought period	SPEI length	Threshold
South Germany	Main vegetation period (MVP)	3 months (SPEI-3)	-1.5 (Severe drought)
Whole Germany	<i>Spring</i>	<i>2 months (SPEI-2)</i>	<i>-1.0 (Moderate drought)</i>
	<i>Summer</i>	<i>4 months (SPEI-4)</i>	

In italic are the variation used only for the robustness checks.

5.5 Results

5.5.1 Main Results

We found that a drought shock, i.e. ‘drought effects’, in South Germany led to a substantial increase in hay prices, up to +13% in month five after the shock (Fig. 3 and Table 2).⁶⁰ The hay price increase lasted from month 3 to 16 after the drought shock (see Figure 3 and Table 2, A4 and A5 for details on other than the 5% significance level). Germany-wide drought shocks resulted in similar effects on hay prices, which peaked +15% and lasted from month 3 to 14 after the drought shock. Differently to this, we found no significant drought effects on feed grain prices, independent if droughts occurred in South Germany or whole Germany.

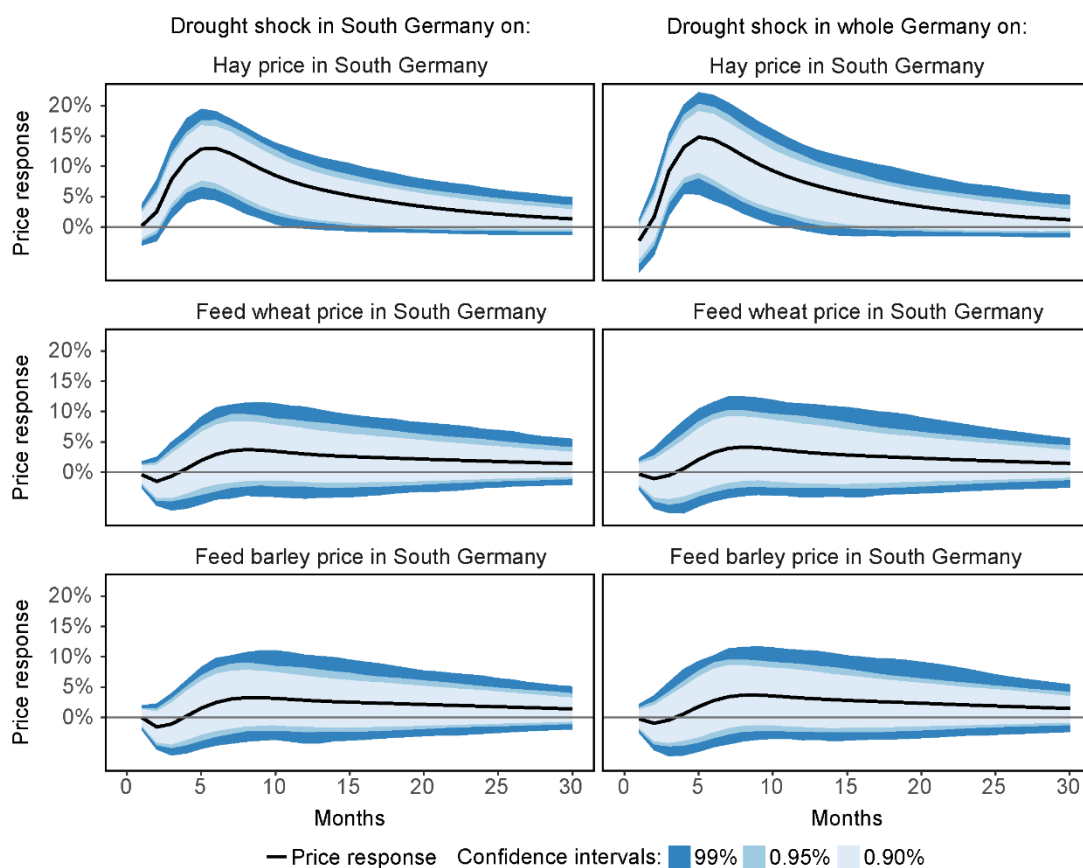


Figure 3: Impulse response functions of the hay, feed wheat and feed barley price in percent to a drought shock (baseline scenario) for South Germany and whole Germany.

⁶⁰ Coefficients estimates are available upon request.

5.5.2 Robustness Checks

In our robustness checks we varied the drought specification with respect to timing of drought, SPEI length and drought threshold (Table 2). Considering only droughts in spring or summer, we found that summer droughts (at regional and national level) caused increases in hay prices. In contrast, we found no effects of spring droughts on hay prices. Drought effects on feed grain prices remained absent in South Germany for spring or summer droughts in almost all cases. On the national level, we also found no generally spring or summer drought effect on feed grain prices (Table 2). When altering SPEI length from SPEI-3 to SPEI-2 or SPEI-4, drought effects on hay prices remained similar. For feed grain prices, we discovered in some cases drought effects when drought specification was based on SPEI-2, whereas for the other SPEI lengths no drought effects were present (Table 2). Decreasing the threshold for drought severity from -1.5 (severe drought) to -1.0 (moderate drought) decreased the magnitude and duration of the drought effects on hay prices. The threshold choice did not impact the drought effects on feed grain prices.⁶¹

⁶¹ Note that results were also similar when droughts are computed for all area of South Germany and Germany and not only for the agricultural area.

Table 2: Drought effects (peak and duration) for different drought specification. *Remark:* Drought effects in South Germany and in whole Germany derived from the impulse response function (Figure 3). %-Numbers indicate the peak effects and numbers in parentheses the start and end month of the effects. We only report values when effects were significant at 5% level (for other significance levels see Table A4 and A5). Grey shaded cells indicate the baseline drought specification and NAs specification without drought observation. We note that results were similar when droughts are computed for all area of South Germany and Germany and not only for the agricultural area.

			SPEI-3		SPEI-2		SPEI-4	
			-1	-1.5	-1	-1.5	-1	-1.5
Droughts in South Germany	Hay price	Main vegetation period	11% (4-14)	13% (3-16)	8% (3-12)	13% (3-14)	12% (4-11)	16% (3-14)
		Spring	-	-	-	-	-	-
		Summer	10% (4-12)	14% (3-14)	10% (3-12)	12% (3-13)	9% (4-8)	15% (4-14)
	Feed wheat price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	8% (1-6)	-	-	-
		Summer	-	-	-	-	-	-
	Feed barley price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	4% (1-2)	2% (1-1)	-	-
		Summer	-	-	-	-	-	-
Droughts in whole Germany	Hay price	Main vegetation period	12% (3-13)	15% (3-14)	8% (4-13)	12% (3-12)	12% (3-12)	17% (3-13)
		Spring	-	-	-	-	-	NA
		Summer	10% (4-11)	15% (3-14)	10% (3-13)	12% (3-12)	10% (4-10)	16% (3-13)
	Feed wheat price	Main vegetation period	-	-	-	6% (9-9)	-	-
		Spring	-	-	-	-	-	NA
		Summer	-	-	-	-	-	-
	Feed barley price	Main vegetation period	-	-	-	6% (8-10)	-	-
		Spring	-	-	2% (1-1)	-	-	NA
		Summer	-	-	-	-	-	-

5.6 Discussion & Conclusion

We showed that droughts at the regional and national level caused substantial increases in hay prices (up to +15%), while feed grain prices were, in our case study, not affected by droughts. This indicates that feed grain markets are – in contrast to hay markets – organized at higher than regional or national levels and thus react less to regional or national drought shocks. These responses confirm with our theoretical and market assumptions, i.e. that prices of markets with relatively low market integration due to high transport and transaction costs respond stronger to drought shocks. Furthermore, hay prices did not react immediately to droughts, but drought responses occurred with a delay (about three months), and drought-induced price shocks were long lasting (usually for over a year). These observations are in line with our theoretical model and the assumption of relatively low transparency of the hay market. Therefore, our analysis highlights the importance of considering transport and transaction costs with respect to their value to understand the price sensitivity to regional shocks, such as droughts. In general regional and national droughts were highly correlated, which is in line with the systemic nature of droughts and explains similar reaction to regional and national droughts. Climate change will increase to occurrence probability and magnitude of droughts. The here identified price sensitivity of the hay market represents an additional severe risk to the agricultural and livestock sector, next to the risk of yield loss. Farmers may suffer from low feed production and exceptionally high prices for the additional feed bought. Similar argumentation about responses to droughts can also hold true for other markets with low-value-to-weight products, low market transparency, low trade quantities and/or with a lack of formal market exchanges, and particularly for agricultural markets in developing countries that often exhibit high national and international trade costs, i.e. transport and transaction costs, thus, low market integration (Porteous 2019). The knowledge about feed price responses to droughts is important to include into farm management and policy actions, especially under future climatic scenarios. Here, for example, online feed price exchanges might contribute to reduce price shocks as they increase market transparency.

Droughts based on SPEI cover important events of low precipitation and high temperature, which together increase intensity of droughts and often occur together (Trenberth and Shea 2005; Estrella and Menzel 2013). Next to these events also other extreme weather events, as solely extreme high/low temperature and precipitation as well as other interactions than high temperature and low precipitation might be important (e.g. Rosenzweig et al. 2002; Schlenker and Roberts 2009; Barlow et al. 2015; Tack et al. 2017) for feed and other agricultural prices and remain an important area for future research.

5.7 Data Availability

The data that support the findings of this study are openly available at Schaub and Finger (2019; <https://doi.org/10.3929/ethz-b-000385361>).

5.8 Code Availability

The R-code for replication of this study is online available.⁶²

5.9 Acknowledgements

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⁶² See Schaub, S. & Finger, R. (2020). Drought effects on hay and feed grain prices. *Environmental Research Letters*, **15**, 034014. <https://doi.org/10.1088/1748-9326/ab68ab>

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5.11 Appendix 1 – Model Identification

We impose the following restriction on our model ('canonical form'):

$$\begin{bmatrix} u_t^k \\ u_t^{wheat} \\ u_t^{barley} \\ u_t^{hay} \end{bmatrix} = \begin{bmatrix} 1 & 0 & 0 & 0 \\ \alpha_{wheat,k} & 1 & 0 & 0 \\ \alpha_{barley,k} & \alpha_{barley,wheat} & 1 & 0 \\ \alpha_{hay,k} & \alpha_{hay,wheat} & \alpha_{hay,barley} & 1 \end{bmatrix} \begin{bmatrix} \varepsilon_t^k \\ \varepsilon_t^{wheat} \\ \varepsilon_t^{barley} \\ \varepsilon_t^{hay} \end{bmatrix} \quad (A.1)$$

In this restriction specification, i) droughts instantaneous affect all three prices, ii) feed wheat prices affect instantaneous feed barley prices and hay prices and iii) barley prices affect instantaneous hay prices. The restrictions are based on the following assumptions. First, the feed wheat and barley market are more transparent than hay prices, thus, these market would first show trends which are taken up by the hay market. Second, of the total livestock feed in Germany a larger share is wheat than barley (BLE 2019).

5.12 Appendix 2 – Tables and Figures

Table A1: Summary statistics.

	Real prices			Natural long transformed real prices		
	Mean	Median	SD	Mean	Median	SD
Hay price	10.50	9.96	2.88	2.32	2.30	0.25
Feed wheat price	15.14	14.47	4.10	2.68	2.67	0.26
Feed barley price	14.74	13.70	3.86	2.66	2.62	0.25

SD refers to standard deviation.

Table A2: Augmented Dickey–Fuller unit root test.

	τ_3	ϕ_2 (constant)
Hay price	-2.9032**	4.3238*
Feed wheat price	-2.7975*	3.9473 *
Feed barley price	-2.875*	4.1634*
Droughts	***	

*, **, *** denote significance at the 10%, 5% and 1% level, respectively. τ_3 is the test statistic for unit root. Null hypothesis is presence of an unit root. For critical values see Table A3. Note that we do not provide test statistics for each drought variable, but all drought variables are significant at a 1% level.

Table A3: Critical values for the Augmented Dickey–Fuller unit root test.

	With a constant		Without a constant
	τ_3	ϕ_2 (constant)	τ_3
1pct	-3.46	6.52	-2.58
5pct	-2.88	4.63	-1.95
10pct	-2.57	3.81	-1.62

Table A4: Drought effects (peak and duration) for different drought specification considering the 10% significance level. *Remark:* Drought effects in South Germany and in whole Germany derived from the impulse response function. Numbers indicate the peak effects and numbers in parentheses the start and end month of the effects. We only report values when effects were significant at 10% level. Grey shaded cells indicate the baseline drought specification and NAs specification without drought observation. We note that results were similar when droughts are computed for all area of South Germany and Germany and not only for the agricultural area.

			SPEI-3		SPEI-2		SPEI-4	
			-1	-1.5	-1	-1.5	-1	-1.5
Droughts in South Germany	Hay price	Main vegetation period	11% (4-17)	13% (3-18)	8% (3-15)	13% (3-16)	12% (3-12)	16% (3-16)
		Spring	-	-	-	-	-	-
		Summer	10% (3-14)	14% (3-16)	10% (3-14)	12% (3-14)	9% (4-12)	15% (3-16)
	Feed wheat price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	8% (1-8)	-	-	-
		Summer	-	-	-	-	-	-
	Feed barley price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	7% (1-7)	5% (1-2)	-	-
		Summer	-	-	-	-	-	-
Droughts in whole Germany	Hay price	Main vegetation period	12% (3-16)	15% (3-17)	8% (3-15)	12% (3-14)	12% (3-17)	17% (3-16)
		Spring	-	-	-	-	-	NA
		Summer	10% (3-13)	15% (3-17)	10% (3-15)	12% (3-13)	10% (4-13)	16% (3-15)
	Feed wheat price	Main vegetation period	-	-	-	6% (7-13)	-	-
		Spring	-	-	-	-	-	NA
		Summer	-	-	-	6% (6-10)	-	6% (7-8)
	Feed barley price	Main vegetation period	-	-	-	6% (6-14)	-	-
		Spring	3% (1-1)	-	3% (1-2)	-	-	NA
		Summer	-	-	-	5% (7-10)	-	-

Table A5: Drought effects (peak and duration) for different drought specification considering the 1% significance level. *Remark:* Drought effects in South Germany and in whole Germany derived from the impulse response function. Numbers indicate the peak effects and numbers in parentheses the start and end month of the effects. We only report values when effects were significant at 1% level. Grey shaded cells indicate the baseline drought specification and NAs specification without drought observation. We note that results were similar when droughts are computed for all area of South Germany and Germany and not only for the agricultural area.

			SPEI-3		SPEI-2		SPEI-4	
			-1	-1.5	-1	-1.5	-1	-1.5
Droughts in South Germany	Hay price	Main vegetation period	11% (4-10)	13% (3-10)	8% (3-9)	13% (3-12)	12% (5-10)	16% (3-11)
		Spring	-	-	-	-	-	-
		Summer	10% (4-8)	14% (3-11)	10% (4-9)	12% (3-10)	-	15% (4-10)
	Feed wheat price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	-	-	-	-
		Summer	-	-	-	-	-	-
	Feed barley price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	-	-	-	-
		Summer	-	-	-	-	-	-
Droughts in whole Germany	Hay price	Main vegetation period	12% (4-9)	15% (3-10)	8% (4-9)	12% (3-10)	12% (4-8)	17% (3-9)
		Spring	-	-	-	-	-	NA
		Summer	10% (4-7)	15% (3-11)	10% (4-10)	12% (3-8)	10% (4-6)	16% (3-10)
	Feed wheat price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	-	-	-	NA
		Summer	-	-	-	-	-	-
	Feed barley price	Main vegetation period	-	-	-	-	-	-
		Spring	-	-	-	-	-	NA
		Summer	-	-	-	-	-	-

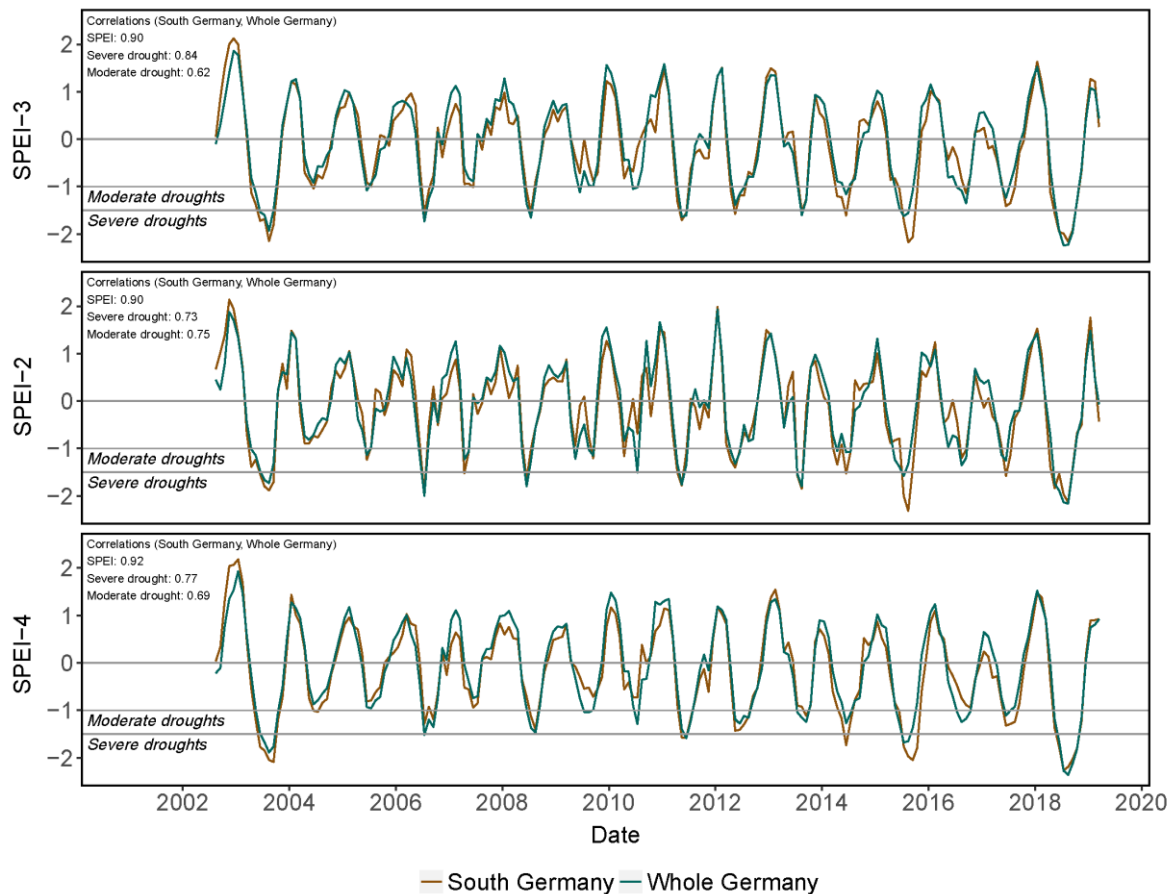


Figure A1: SPEI-3, SPEI-2 and SPEI-4 between August 2002 and April 2019. The two grey lines at -1.5 and -1 represent the threshold for severe and moderate droughts, respectively. On the top left of each panel the correlation between the SPEI, severe droughts and moderate droughts in South Germany and whole Germany are indicated. Correlation refers to the Pearson correlation coefficient. We used only droughts during the main vegetation period for calculating the correlation of droughts.

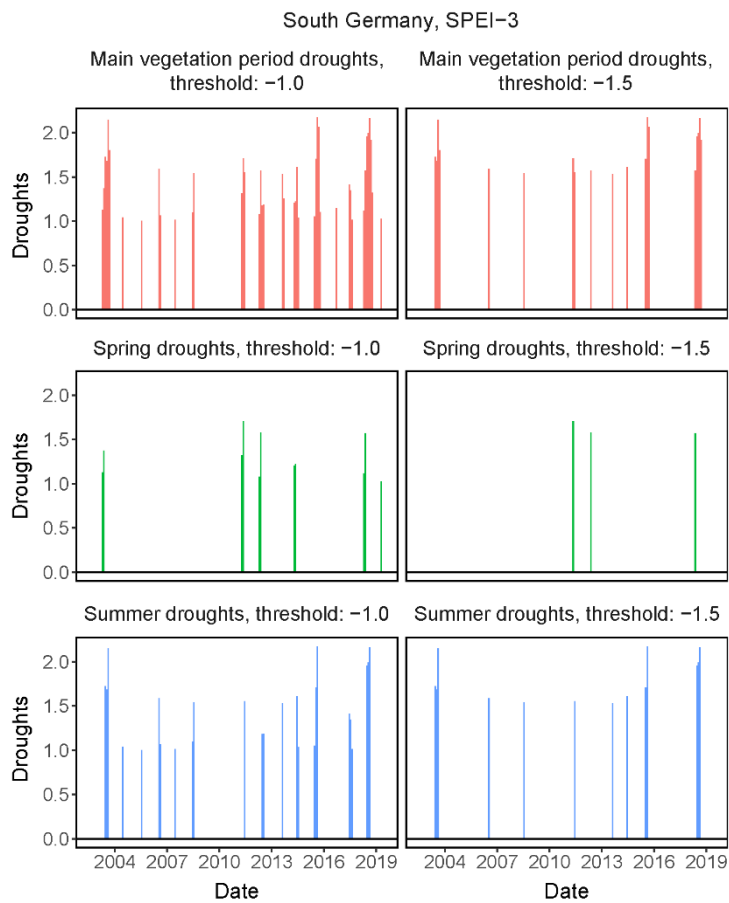


Figure A2: Droughts in South Germany for SPEI-3 and different drought specifications.

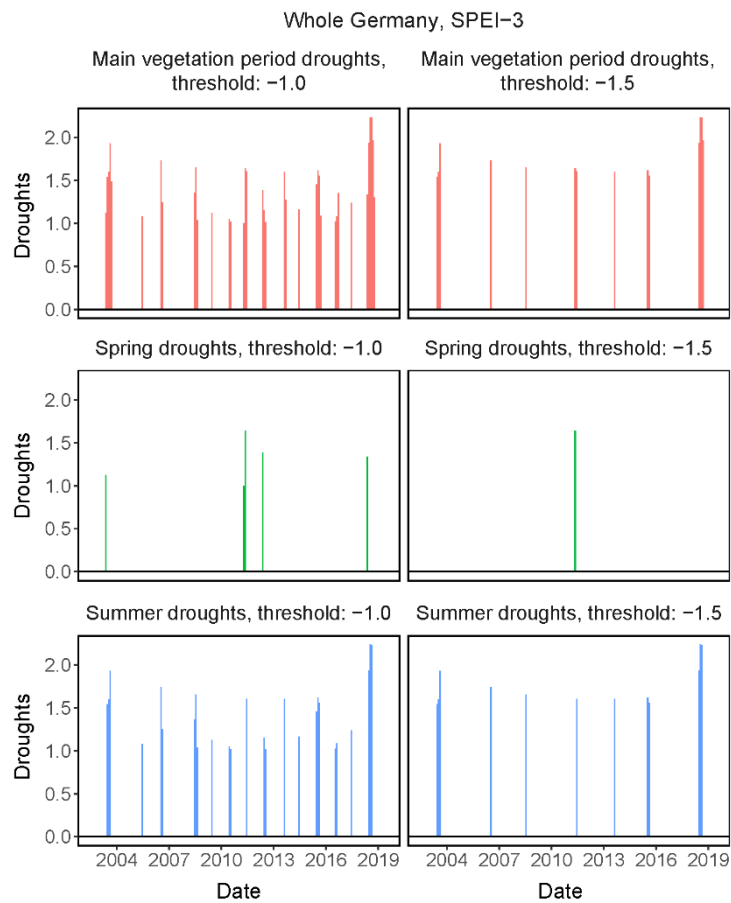


Figure A3: Droughts in whole Germany for SPEI-3 and different drought specifications.

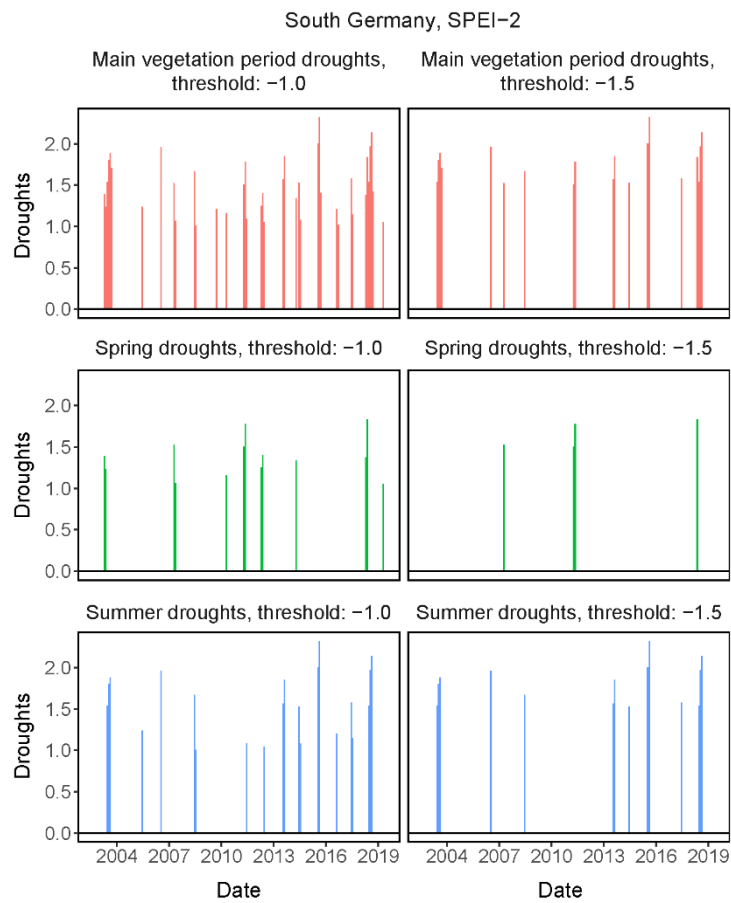


Figure A4: Droughts in South Germany for SPEI-2 and different drought specifications.

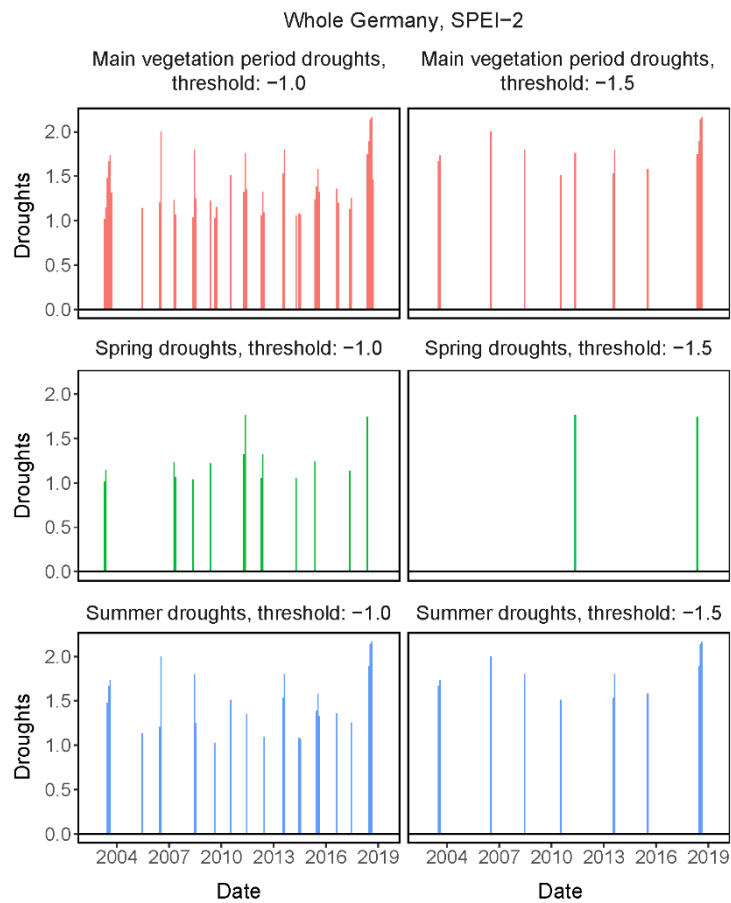


Figure A5: Droughts in whole Germany for SPEI-2 and different drought specifications.

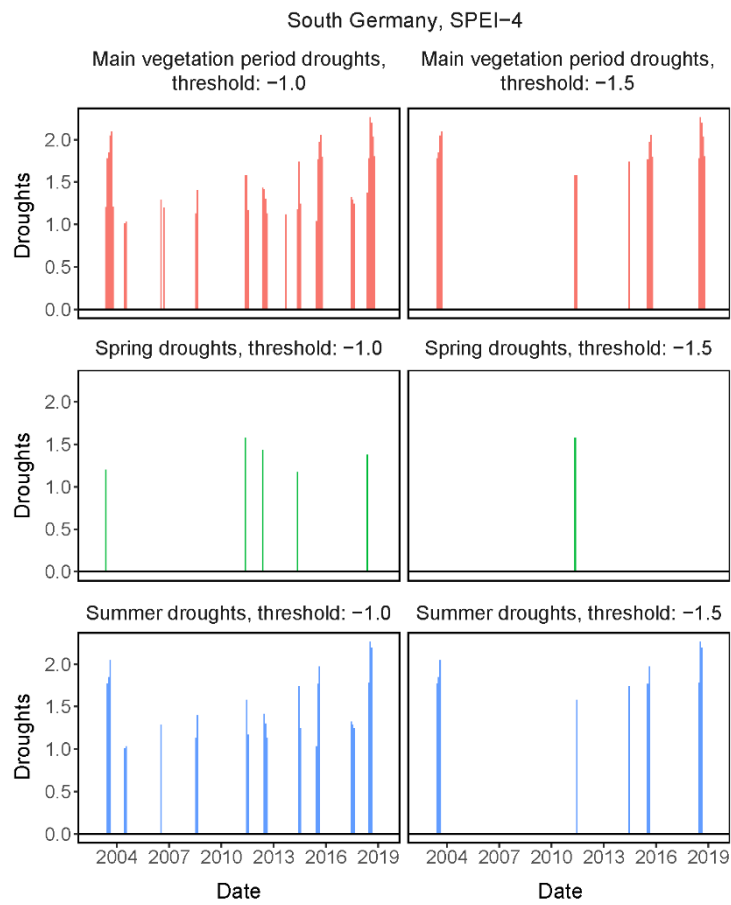


Figure A6: Droughts in South Germany for SPEI-4 and different drought specifications.

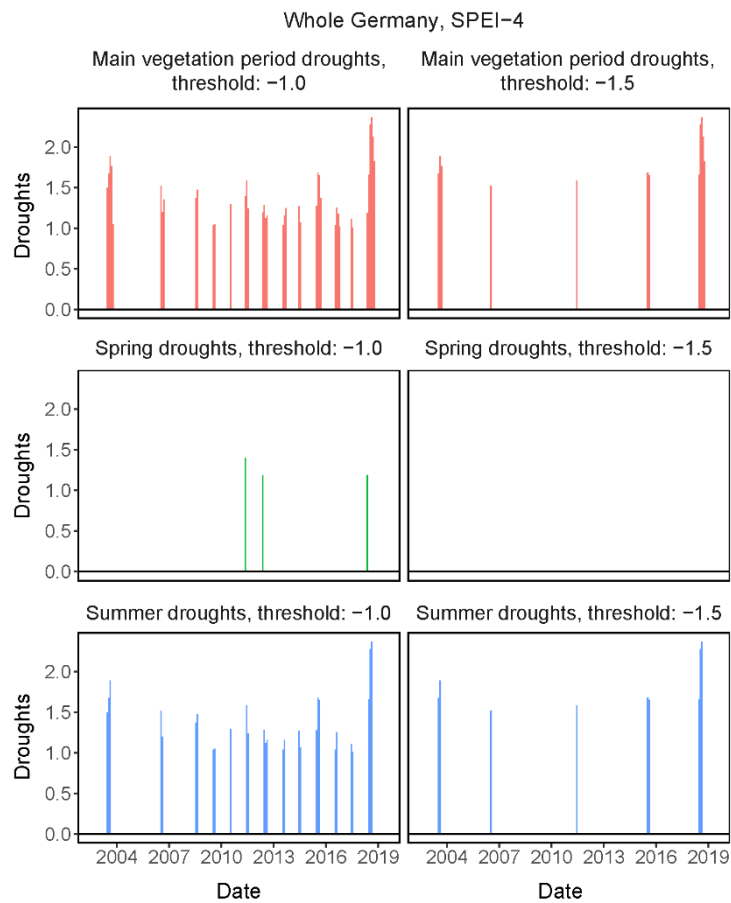


Figure A7: Droughts in whole Germany for SPEI-4 and different drought specifications.

6 Thesis Appendix 1 – [A Way Of] Exploring our Tolerance Towards Biodiversity Loss and Re-thinking our Biodiversity Communication

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Abstract

We experience a threatening biodiversity loss, however, we remain (quite) tolerant to it. I explore this phenomenon from a behavioral economic perspective and think about how we can improve communication about biodiversity loss. From this, we learn that: 1) It matters where we set our expected biodiversity value, because depending on it we experience biodiversity loss as either a loss event or simply less of a gain. 2) The value of biodiversity has a certain and an uncertain dimension. Too often, we ignore the uncertain aspects of biodiversity, which has an increasing value for future generations. 3) We need to connect biodiversity at large to findings and discoveries we make thanks to single species. 4) Lastly, emphasizing the many different values lost by losing biodiversity can improve biodiversity loss communication. Applying these ideas can improve communication of biodiversity loss, decreasing the tolerance towards it.

Keywords: biodiversity loss, communication, behavioral economics, prospect theory, biodiversity values

6.1 Essay

We, humans, and nature are undoubtedly interlinked. We affect nature and the rich biodiversity it provides us on an accelerating and threatening speed (IPBES 2019). When the IPBES 2019 report was released and the news spread (again) about the unprecedented biodiversity loss, it was clear that the majority of people, at least within my experience, had a strong consensus on acting towards a more sustainable development direction. So, why are we experiencing such unrestrained development and how can we improve communication to reduce (our tolerance towards) biodiversity loss? I began reflecting upon these questions from a behavioral economic perspective:

Biodiversity describes the diversity within species, between species and of ecosystems (IPBES 2019). The complexity and magnitude of biodiversity is impressive and amazing, while at the same time making it a difficult subject for valuation. The biodiversity values that come to mind are in the form of ecosystem services, which are all direct and indirect benefits we derive from ecosystems.

In behavioral economics, we need to know our reference point (what we expect) in order to value outcomes and understand how we experience events of losses and gains (Kahneman and Tversky 1979, Thaler 1985). The reference point does not have to be zero. For example, if your next paycheck is 1900€ instead of 2000€ you experience a loss event even though you still receive money, this is because the reference point for your next paycheck was 2000€. After we set our reference point, every outcome that is lower/higher is a value loss/gain. Furthermore, we experience gains and losses differently. This simply means that when we lose 100€, we have a bigger value loss than a value gain when we gain 100€. Additionally, when we lose 200€ at once, we experience less of a value loss than losing twice 100€, which is likewise for gains. These considerations constitute the so-called *value function* (Fig. 1).

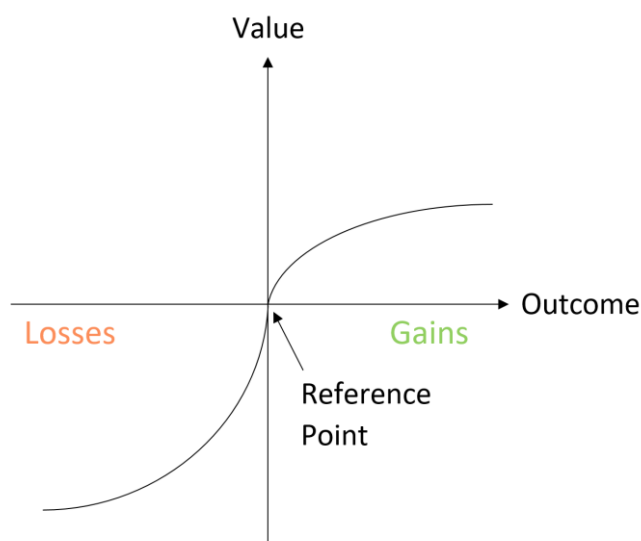


Figure 1: Value function from prospect theory.

In order to fit biodiversity to this function, we need to assign values to it. The biodiversity value originates out of two ‘worlds’: the certain and the uncertain world (Bartkowski 2017). Values from both worlds are highly important when setting the reference value and need to be considered under a societal rather than individual perspective – keeping also future generations in mind. In the *certain world*, are all the biodiversity values that we derive from the influence of biodiversity on ecosystem services (provisioning, cultural, supporting and regulating, Fig. 2). The *uncertain world* is extremely complex and its real values are not known. These values encompass the insurance, option and spillover values (Bartkowski 2017). More specifically, first, biodiversity provides us with an **insurance value**, when it leads to more stable provision of ecosystem services. For example, plant diversity increases stability of hay yields in grasslands (Finger and Buchmann 2015). Within the insurance value, I favor to highlight additionally the **resilience value**. It emphasizes the function of biodiversity to avoid the collapse of a system, as observed, for example, in marine ecosystems (Worm 2006). Second, maintaining a higher biodiversity retains the option for possible future uses, the so-called **option value**. This can be a substantial value, just consider that about 70% of cancer medicines would not be known without nature’s diverse species pool (IPBES 2019). Third, the **spillover value** relates the biodiversity contribution to interaction of ecosystems (services) in space. This can be observed with migratory species, such as salmon and cranes, which are depending on different ecosystems to persist. Note that the biodiversity value from the uncertain world increases with time, thus, when we consider future generations.

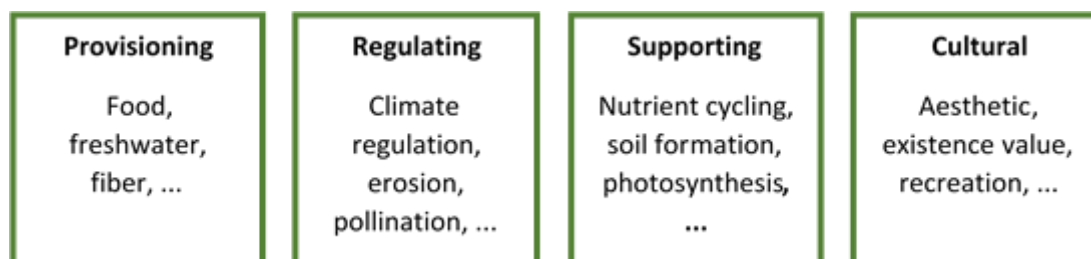


Figure 2: The four different types of ecosystem services. Source: Millennium Ecosystem Assessment 2005. *Note:* These ecosystem services include for example that the amazing diversity of fruits and the many different pollinators that ensure these fruits, the beauty of scuba diving in diverse underwater environment or the simple knowledge of the existence of such a rich biodiversity.

Based on how we experiences losses and gains as well as the biodiversity values, I recommend the following for reducing tolerance towards biodiversity loss and improving the tone and efficacy of biodiversity loss communication:

First, it is important to re-calibrate the reference point using all biodiversity values (i.e. certain and uncertain) because **it matters how we experience biodiversity loss**. Depending where our reference

value is, biodiversity loss can be experienced as a loss event or just an event of less of a gain. This can change considerably how we value the situation. Therefore, the re-calibration of the reference point might bring forward higher willingness to act against biodiversity loss and to bear higher conservation costs.

Second, a **large part of the biodiversity value is uncertain**. Which is often enough not considered in the valuation of biodiversity. However, these values have tremendous potential and ignoring them is not acceptable. Acting precautionary would be much more reasonable. Moreover, we face an ever **faster changing world**, and especially in this situation, biodiversity can help tackling the unforeseen challenges.

Third, we need to connect the value we gain from single species not (only) to the single species but to biodiversity at large. Just think about it, the chance of finding one species that help us to advance in medicine, or also in other fields, in a large pool of species is simply higher than pursuing this task in a pool of few species. A large biodiversity pool provides us with a **rich toolset** to tackle existing and future challenges. Therefore, we need to include this connection in the biodiversity valuation and communicate it clearly.

Fourth, given a reference point we need to think about how to communicate biodiversity losses (and gains). Considering the subjective experience of losses (and gains), it would be a good strategy to **segregate biodiversity loss** (and gains) in our communication. Hence, informing about many different values and facets of biodiversity that are lost. In the case of the IPBES report, it might not be efficient to report every single one of the 1 million species facing extinction separately, but to clearly emphasize the different biodiversity facets and (certain and uncertain) values that can be lost.

Systematically acknowledging and incorporating all these concepts into biodiversity loss communications and policy recommendations can help to gather support for biodiversity conservation and set directions for more sustainable policies.

6.2 Acknowledgements

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7 Thesis Appendix 2 – Global Climate Change and Biodiversity Interest Trends, Research-Policy Efforts and Activism

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Abstract

Climate change and biodiversity loss threaten our life as we know it. This caused considerable attention by researchers, policy makers and the civil society, leading to major research-policy efforts (including IPCC and IPBES reports) and activism (including global climate strikes). Here, we investigate how regional and national public interest changed over time (2011-2019) and responded to research-policy efforts and activism. We find that in many countries public climate change interest increased over time, especially when considering the more recent years (2016-2019). In contrast, for most countries we do not observe such positive trends for public biodiversity interest. The relationship between public interest and research-policy efforts or activism were heterogeneous in time and between countries. However, we hypothesize that in many countries research-policy efforts and activism contributed directly or indirectly to increased climate change interest in more recent years. Our study highlights the importance of considering differences between countries in the climate change and biodiversity loss communicating and when designing policies as well as that, especially for biodiversity, we need to upgrade communication.

Keywords: public interest, IPCC, IPBES, climate strikes, google trends, environmental engagement.

7.1 Introduction

Climate change and biodiversity loss threaten our life as we know it (IPCC 2018, IPBES 2019). These threats are increasingly recognized by economic actors (Granados Franco et al. 2020). The topics of climate change and biodiversity also attract major attention by researchers, cause policy discussion and activism (Legagneux et al. 2018, Knutti 2019, Granados Franco et al. 2020). Major outcomes of research and policy efforts are the intergovernmental reports by the Intergovernmental Panel on Climate Change (IPCC) and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). The activism and concerns about climate change and biodiversity by a part of the public led to (global) climate strikes. We investigate how regional and national public interest changed over time and responded to research-policy efforts and activism.

How events shape the public interest in climate change and biodiversity over time and how these are connected to regional differences is however little understood. Especially, open questions remain about more recent changes in trends (i.e. since about the sixth IPCC report) and how research and policy efforts as well as activism affected these trends. This knowledge, and knowing regional differences, is crucial for understanding public policy support as well as for improving communication of these topics and designing effective policies.

The interrelationship between climate change and biodiversity (loss) makes a joint analysis of public interest in these topics very important. Two aspects are particularly important in this regard: First, climate change leads to biodiversity loss and biodiversity can help mitigate adverse climate change effects (Chapin et al. 2000, Thomas et al. 2004, Isbell et al. 2015, Warren et al. 2018, Trisos et al. 2020). Second, the topics can compete for public interest, which is especially relevant as climate change is often better understood by individual members of the public and considered to be more of an intermediate threat to our current living standards than biodiversity loss (Zaccai and Adams 2012, Legagneux et al. 2018).

Studies investigating trends of interest in climate change and/or biodiversity report ambiguous findings (Mccallum and Bury 2013, Anderegg and Goldsmith 2014, Funk et al. 2014, Proulx et al. 2014, Nghiem et al. 2016, Troumbis 2017). Ambiguous results for biodiversity interest were also reported in the more recent past, while climate change interest, at least in the US, increased (Chevallier et al. 2019, Troumbis 2019). Studies that investigated events, e.g. activism or scandals, reported a positive correlation between events and interest in some but not in all cases (Anderegg and Goldsmith 2014, Cooper et al. 2019, Mavrodieva et al. 2019). There is a lack of global studies that test trends over time and correlations with research-policy efforts and activism across regions and countries. This is despite that public interest in climate change and biodiversity interests can be linked to cultural differences, climate vulnerability or economic and institutional development differences (Funk et al. 2014, Archibald and Butt 2018, Baynham-Herd et al. 2018, Chevallier et al. 2019, Cooper et al. 2019), which

makes a global study with a regional and national explicit focus that includes recent years extremely important.

We contribute on closing these gaps by adding an empirical as well as regionally and nationally differentiated perspective on recent global public interest trends in climate change and biodiversity and on how these trends related to research-policy efforts and activism. To empirically identify trends in climate change and biodiversity interest and regional differences in these trends, we leverage Google search data of 200 countries between 2011 and 2019. Google is by far the most commonly used search engine and Google search data provides a large sample that is a commonly used for measure for public interest and engagement in a topic (Ripberger 2011, Nghiem et al. 2016, Troumbis 2017, Burivalova et al. 2018, StatCounter 2020). We empirically test trends on the regional and national scale and whether long-term (2011-2019) and medium-term (2016-2019) trends differ. Moreover, we analyse how research-policy efforts (in terms of intergovernmental reports) and activism (in terms of climate strikes) influenced public interest in climate change and biodiversity. Within the analysis of the interest response to the research-policy efforts, we focus on three out of five reports: the IPCC reports (AR5) and IPCC report (Global Warming) for climate change interest and IPBES report (Global Assessment) for biodiversity interest as the former two are land mark reports for climate change and the latter for biodiversity. For activism, we focus on the first and the third global climate strikes in our study period (out of a total of four global climate strikes). These were the largest strike in terms of participants (FridaysForFuture 2020) and the third global climate strikes, different to the other strikes, lasted over a week and the biodiversity topic was more evident.

We find that in many countries interest in climate change increased over time, especially when considering the more recent past (2016-2019). In contrast, for biodiversity interest we do not find such positive trends in most countries. We do not find a general relationship between public interest and research-policy efforts and activism. However, we hypothesis that in countries with positive climate change interest trends in the recent past all the research-policy efforts and climate strikes contributed directly or indirectly to increased climate change interest.

7.2 Results

7.2.1 Interests Over Time

We find that public climate change interest globally and across regions generally decreased in the beginning of the 2010ies, remained rather constant in its middle and increased at the end of it (Fig. 1A). The direction of long-term (2011-2019) trends differ between countries (Fig. 1B1). In many countries climate change interest increased in the long-term, including especially countries in North America, western South America, Europe and China. In contrast, countries in East Africa, West Africa and eastern South America showed more often decreasing long-term trends. Medium-term (2016-

2019) climate change interest trends were generally positive or neutral (Fig. 1B3). Moreover, many countries in the Northern Hemisphere as well as in western South America and Australia showed a substantial positive medium-term trend. For example in Australia, Chile, Germany and USA interest in climate change increased in 2019 compared to 2016 on average by about 36%, 60%, 66% and 31%.

Public interest in biodiversity generally decreased globally and in many regions in the beginning of the 2010ies and remained rather constant since the middle of it (Fig. 1A). Considering national long-term trends, we find that in many western South American, African and Asian countries biodiversity interest decreased (Fig. 1B2). Few countries exhibited a positive long-term trend, amongst which are the USA, China, Saudi Arabia and Turkey. We find that most of the long-term trends disappeared in the medium-term and that countries mostly showed neither a positive nor a negative medium-term trend (Fig. 1B4).

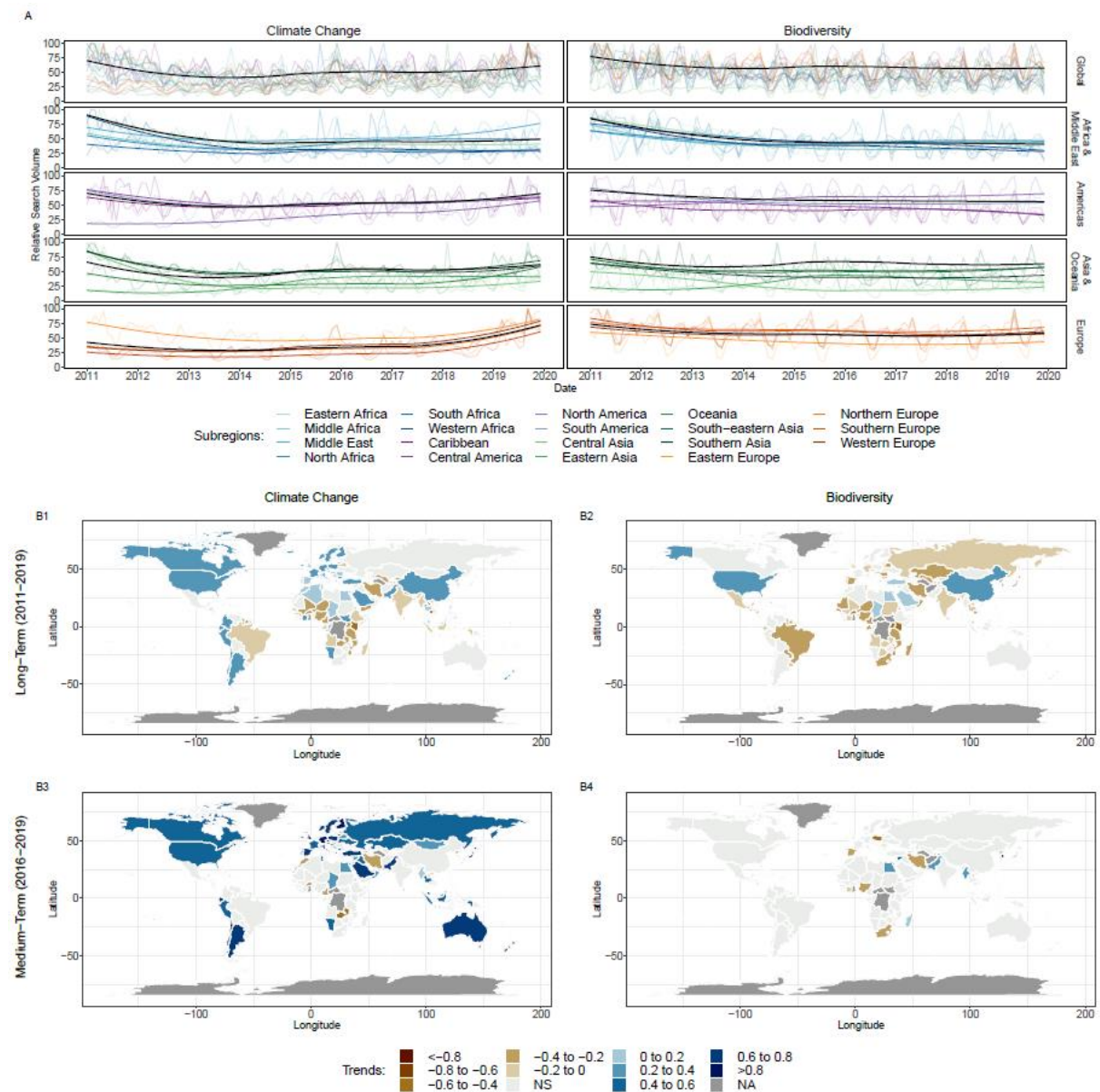


Figure 1: Global, regional and national trends in climate change and biodiversity interest. Public interest is measured by the relative Google search volume. The left section of panel A shows regional

trends of climate change interest and the right section for biodiversity interest. The black lines in panel A represent the aggregated trends of the panel region, i.e. either of the world, Africa & Middle East, Americas, Asia & Oceania or Europe. The non-transparent lines are the trends and the transparent lines the public interest over time of the 19 sub-regions. All trend lines are based on non-parametric local polynomial regression fitting and country level interests are weighted by their population share in the region. Panel B shows national trends for: long-term and climate change (B1), long-term and biodiversity (B2), medium-term and climate change (B3) and medium-term and biodiversity (B4). Trends are indicated as not significant (NS) using the $P > 0.05$ threshold. Standard errors of national trends are corrected for heteroscedasticity and serial-correlation using (HAC) standard errors.

7.2.2 Interests & Research-Policy Efforts

We find heterogeneous relationships of the research-policy efforts and changes in climate change and biodiversity interest trends. The relationships differed between regions and time, i.e. reports (Fig. 2 and Fig. A1 to A3). When considering instead of a 1-week-perspective a 4-week-perspective changes generally are more homogenous across countries (Fig. 3). Moreover, worldwide national level changes are generally very heterogeneous between countries and the global weighted averages changes are usually relatively low (Fig. 3 and Fig. A4 to A7).

In the week after the IPCC report (AR5) release, climate change interest trends increased in the Americas ($P=0.01$; Fig. 2). We observe this trend change also when considering four weeks after the report release ($P < 0.0002$). Considering individual countries, we find that national interest level changes are quite heterogeneous (Fig. 3). Most homogeneously positive are responses in the Americas.

Climate change interest trend increased after the IPCC report (Global Warming) release in the Americas ($P=0.02$) and in Europe ($P=0.002$). In contrast to the IPCC report (AR5) release, the change after the IPCC report (Global Warming) release only persisted in Europe ($P=0.06$) and not in the Americas when considering a 4-week-perspective. Our results show also that in the Americas and Europe some months after the report release interest in climate change again increased at a higher rate. National interest level changes are generally higher in Europe and North America while changes in countries in other regions are heterogeneous (Fig. 3). Notably is that changes in climate change interest in Europe were much more homogenous after the IPCC report (Global Warming) than they were after the IPCC report (AR5) release.

For biodiversity interest, the IPBES report (Global Assessment) release came with a positive trend change in Africa & Middle East ($P=0.01$), the Americas ($P=0.09$) and Europe ($P < 0.0001$). In a 4-week-perspective this is only observed for Europe ($P < 0.0001$). National interest level changes were again not clustered by regions, except for Europe that showed generally positive national interest level changes (Fig. 3).

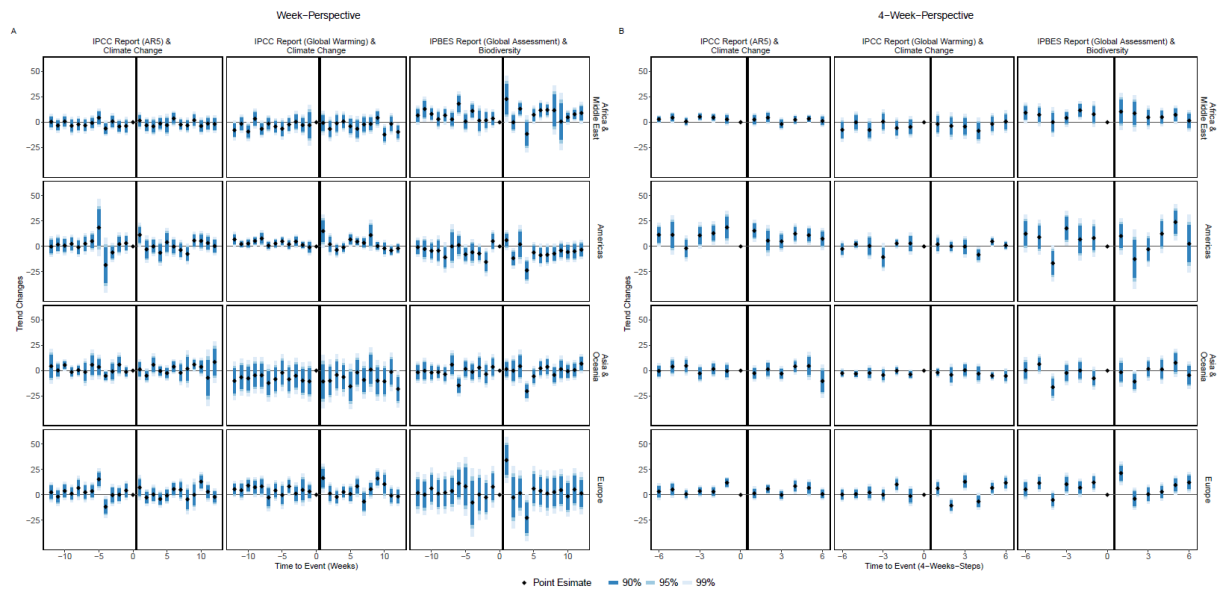


Figure 2: Regional trend changes in climate change and biodiversity interest around the research-policy efforts. Public interest is measured by the relative Google search volume. The panel A shows weekly trend changes and the panel B four-weekly trend changes. We show in this figure changes in a) climate change interest trends for the IPCC report (AR5) release in November 2014 and the IPCC report (Global Warming) release in October 2018 and b) biodiversity interest trends for the IPBES report (Global Assessment) release in May 2019. The diamonds are point estimates and the bars show the 90%, 95% and 99% confidence intervals. Point estimates and confidence intervals are from a population weighted linear regression model. We correct standard errors by clustering over nations and time.

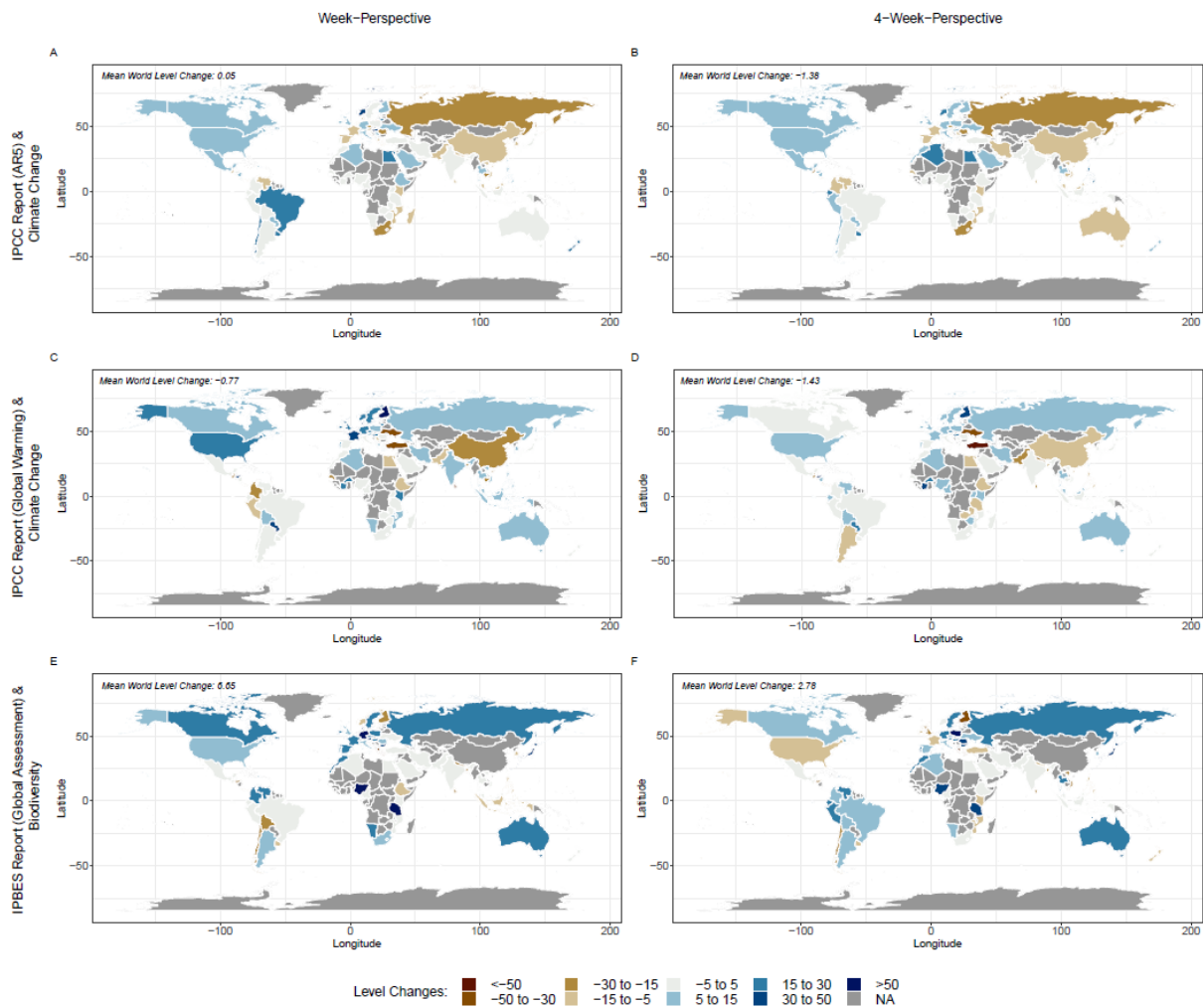


Figure 3: National interest level changes in climate change and biodiversity in the week or four-week period after a research-policy effort compared to the week or four-week period in which a research-policy effort took place. Public interest is measured by the relative Google search volume. The left panel shows weekly changes and the right panel four-weekly changes in interest levels. We show in this figure changes in a) climate change interest level for the IPCC report (AR5) release in November 2014 (A and B) and the IPCC report (Global Warming) release in October 2018 (C and D) and b) biodiversity interest level for the IPBES report (Global Assessment) release in May 2019 (E and F). The mean world level change is weighted by population.

7.2.3 Interests & Activism

The changes in climate change and biodiversity interest trends after activism, in terms of global climate strikes, were heterogeneous between regions and strikes (Fig. 4 and Fig. A8 to A10). Observed climate change and biodiversity interest trend changes in a week perspective were often not observed in a four week perspective. Moreover, nations without activism (mostly countries in Africa & Middle East) also showed heterogeneous interest trend changes after strikes. We do not find a general relationship between changes in climate change and biodiversity interest after a strike. Considering only national

climate change interest level changes after a global strike, we find that when national interest levels increased after a strike, these increases were much higher than national increases after research-policy efforts (Fig. 5 and Fig. A11 to A14). In parallel, the world average climate change interest change, when interest increased, were usually considerably higher after a strike than after a research-policy effort. The week after the first global climate strike (Global Climate Strike (1)) interest in climate change increased in the Americas ($P < 0.0001$) and Europe ($P < 0.0001$); as well as in nations without activism in Africa & Middle East ($P = 0.001$; Fig. 4A). These changes could not be observed when considering a four-week-perspective (Fig. 4B). Nevertheless, on a national perspective, we find many countries with positive change in a one and four week perspective, especially in South America and Europe (Fig. 5). The Global Climate Strike (3) corresponded to an increase in climate change one week after the event interest trend in the Americas ($P = 0.02$), Asia & Oceania ($P = 0.002$) and Europe ($P = 0.001$; Fig. 4A). For Asia & Oceania ($P = 0.001$) and Europe ($P < 0.0001$) the trend changes were also present in a four-week-perspective. The Global Climate Strike (3) corresponded most positively with public climate change interest among all strikes and research-policy efforts (Fig. 5). Following the trend changes, we find that national climate change interest level changes were most commonly observed in countries in the America, Asia & Oceania and Europe, both in a one and four week perspective.

Considering trend changes in biodiversity interest for the Global Climate Strike (3), we find for the week following the strike an increase in Asia & Oceania ($P = 0.007$) and a decrease in Africa & Middle East ($P = 0.04$). Considering a four week perspective, trend changes in Asia & Oceania ($P = 0.08$) are even negative. This picture is also represented when considering national biodiversity interest level changes, as national relationships are rather weak as well as spatially and temporally heterogeneous (Fig. 5).

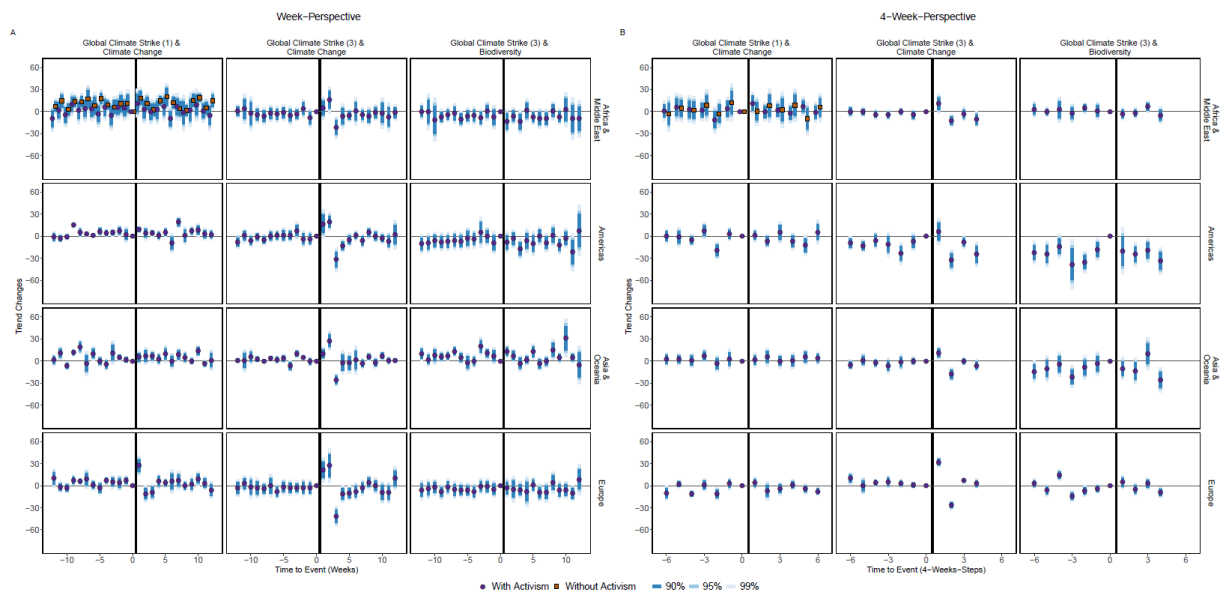


Figure 4: Regional trend changes in climate change and biodiversity interest around activism. Public interest is measured by the relative Google search volume. Changes are indicated for countries with (purple cycles) and without (orange squares) strikes separately. Countries without strikes are only included if at least three countries in a region was without activism. The panel A shows weekly trend changes and the panel B four-weekly trend changes. We show in this figure changes in a) climate change interest trends for the Global Climate Strike (1) in May 2019 and b) climate change and biodiversity interest trends for the Global Climate Strike (3) in September 2019. The circles and squares are point estimates and the bars show the 90%, 95% and 99% confidence intervals. Point estimates and confidence intervals are from a population weighted linear regression model. We correct standard errors by clustering over nations and time.

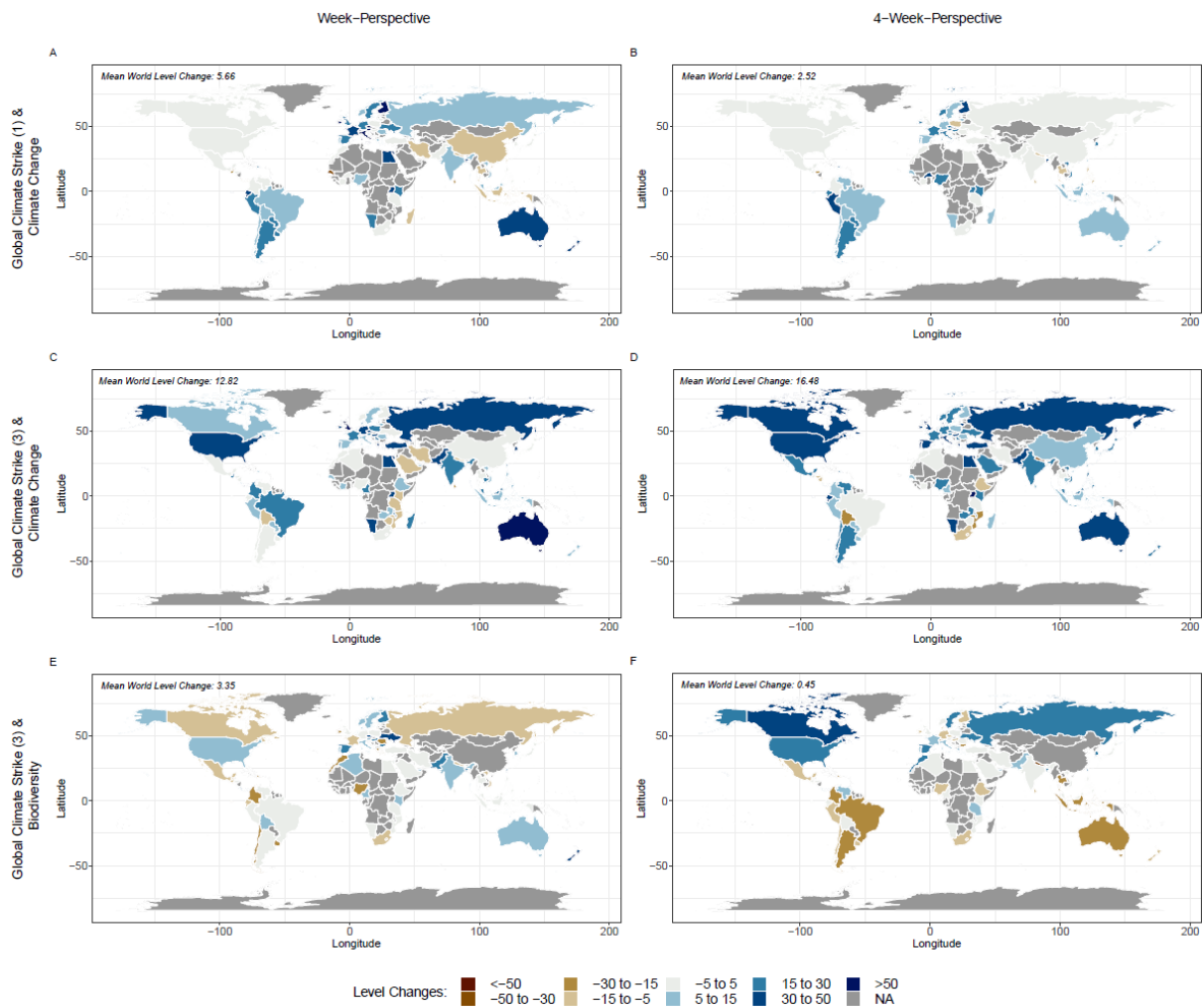


Figure 5: National interest level changes in climate change and biodiversity in the week or four-week period after a strike compared to the week or four-week period in which a strike took place. Public interest is measured by the relative Google search volume. The maps only include countries with strikes. The left panel shows weekly changes and the right panel four-weekly changes in interest levels. We show in this figure changes in a) climate change interest level for the Global Climate Strike (1) in May 2019 (A and B) and b) climate change (C and D) and biodiversity (E and F) interest level for the Global Climate Strike (3) in September 2019. The mean world level change is weighted by population.

7.3 Discussion and Conclusion

We present a global analysis of how public interest in climate change and biodiversity changed over time and how interest is related to research-policy efforts and activism. Our results show that interest in climate change in the medium-term increased in many countries; especially in some countries in the Northern Hemisphere as well as in western South America and Australia medium-term trends were substantial. The long-term trends in climate change interest, were more heterogeneous between countries. The positive medium-term interest trend in climate change is in line with previous findings for the USA (Troumbis 2019). Moreover, similar to earlier studies we find decreasing climate change

interest in many regions in the beginning of the 2010ies (Anderegg and Goldsmith 2014, Proulx et al. 2014). While these studies did not cover more recent years, we note that these trends became positive in many regions at the end of the 2010ies. In contrast to the public interest in climate change, we find that biodiversity interest were constant in most countries in the medium-term and in the long-term even rather negative trends. Constant to decreasing biodiversity interest trends between 2012 and 2017 were also recently reported in a study of eight selected countries (Chevallier et al. 2019). Previously shown positive biodiversity interest trends for the USA in the 2010ies (Troumbis 2019), are, as an exception of our national trends, in line with our long-term findings. However, we do not find positive trends in the USA when considering a medium-term perspective.

Our analysis shows that interest in climate change and biodiversity to research-policy efforts and activism were generally heterogeneous. Previous studies that visually investigated the relationships between aggregated global interest and events (including reports, films or strikes) found as well heterogeneous relationships; among the events in these studies that exhibited positive correlations with interest were the IPCC report (Global Warming), the International Biodiversity Day or the Global Climate March in 2015 (Cooper et al. 2019, Mavrodieva et al. 2019). Furthermore, in the cases when research-policy efforts or activism led to interest increases in our analysis, we find that the increase after strikes were substantially higher. Therefore, strikes seem to more directly correspond to changes in public interest (and search behaviour). This might be partly linked to the collective action of many citizens and scientists taking place in many countries when global strikes happened (Fisher 2019, Warren 2019). However, research-policy efforts and activism shall not be seen in isolation, as for example research-policy efforts, especially the IPCC report (Global Warming), might be linked to the formation of activist movements (Boykoff and Pearman 2019). Furthermore, we note that especially in the end of 2019 many research-policy efforts and activist events took place, which were often temporally close. This makes linking climate change and biodiversity interest changes to single events, especially over longer time, difficult.

Furthermore, we do not find evidence that climate change specific events, such as reports or strikes, drove attention directly away from biodiversity, or vice versa. Previous studies raised concerns about the competition of these topics and suggested such competition trends for media coverage or funding (Veríssimo et al. 2014). We find that while in many countries interest in climate change increased, such large scale developments are not observed for biodiversity. This might be explained as climate change compared to biodiversity is better understood by individuals and is experienced as more an intermediate threat to our current living standards than biodiversity loss, by higher media coverage of climate change related topics as well as better communication strategies (Zaccai and Adams 2012, Veríssimo et al. 2014, Legagneux et al. 2018). However, only increasing media coverage on biodiversity might not be sufficient to increase biodiversity interest (Chevallier et al. 2019)

Technological developments and open access policies offer access to large amount of public interest and opinion data, which covers large shares of the population and long time periods. We exploit these developments by leveraging Google search data of 200 countries between 2011 and 2019. While the data comes with many advantages, it also has certain limitation, e.g. focuses on the population share that uses google or cultural differences in internet use behaviour. Moreover, using this data we observe regional clusters of lower representation (e.g. in Africa or Asia) and that small countries are less represented. This might be linked to less internet users, less interest in these topics or more volatile search behavior in these regions/countries. We note that the regions with lower representation also show more heterogeneity with respect to trends over time and relationships to research-policy efforts and activism. However, using search volume data for representing public interest still allows us covering cost-efficiently far more countries than usually with traditional data collecting approaches.

In conclusion, our findings point towards research-policy efforts and activism contributing in many countries to a positive trend climate change interest in the recent past; despite interest responses to single research-policy efforts and activism being heterogeneous. In contrast, we did not observe the likewise tendencies for public biodiversity interest. The differences in successfully increasing interest between these topics might be linked to differences the public perception of these climate change and biodiversity, media coverage, and/or in communication. The substantially heterogeneities between topics and countries (e.g. due to culture or economic and institutional development) need to be addressed in the communication about climate change and biodiversity loss by researcher, policy-makers, NGOs and other stakeholders as well as in the design of policies. Our study also highlights important areas for future research, including the analysis of which traits drive differences in public interest trend between countries or understanding why public biodiversity interest, despite all the efforts, are not increasing and which communication strategy could be used to tackle the lack of biodiversity interest trend increase.

7.4 Methods

7.4.1 Global Public Interest

For measuring public interest in a topic, we use data on national Google search volume of the topics climate change and biodiversity between 2011 and 2019 from Google Trends. Google summarises in topics all search terms representing the topic in different languages. The search volume per country and per time period downloaded ranges between 0 and 100 and the data is a random sample of all search inquires and it is normalized and rescaled (Google 2020). Google provides only data if search volume in a country was sufficiently high. Thus, countries with generally very low search volume resulted in no data over the entire time period and in other countries that had certain months/weeks

with very low search volume Google provided a score of zero, even when in this month/week some search request were made (Da et al. 2011).

We retrieved monthly as well as weekly search volume data using the R package 'gtrendsR' (Massicotte and Eddelbuettel 2019). We used monthly data for analyzing interest trends over time and weekly data for analyzing the changes in interest trends and levels after research-policy- efforts or activism. For the analysis of research-policy efforts or activism, we needed a high temporal resolution. Therefore, we only regarded national data per country as sufficient when at least 90% of the time values were different from zero (Table A1). In the end, monthly data for climate change interest was available for 200 countries and for biodiversity interest for 178 countries and weekly data for climate change interest was available for 95 countries and for biodiversity interest for 74 countries. Furthermore, to avoid biases in changes around a trend due to seasonal variation in interests, we detrended the weekly data based on years without (the here considered) research-policy efforts or activism. We will provide all code to download the data and to replicate the results. Given that the Google provides different random data samples between downloads, the results can slightly change. The here used data (downloaded on January 22 2020) will be available on request (after publication).

We divided countries in five regions (Africa & Middle East, Americas, Asia & Oceania and Europe) and 19 sub-regions. The regional division is based on World Bank (2018) and United Nations (2020).

7.4.2 Research-Policy Efforts & Activism

We consider in total five research-policy efforts (i.e. intergovernmental reports) and four activist events (i.e. global climate strikes; (Fig. 6). We consulted IPCC and IPBES material to assemble information of the research-policy efforts. In the time period from 2011 to 2019 five intergovernmental reports were released: Four IPCC reports, thus, reports predominantly linked to climate change, and one IPBES report, thus, a report predominantly linked to biodiversity. Within the research-policy efforts, we especially focus on the IPCC reports (AR5) and IPCC report (Global Warming) for climate change interest and IPBES report (Global Assessment) for biodiversity interest as the former two are land mark reports for climate change and the latter for biodiversity. Additionally, in previous years to those IPCC reports no other IPCC reports were released and likewise for the IPBES report. For activism we restricted the strikes to global climate strikes between 2011 and 2019 and those listed by FridaysForFuture (2020). This resulted in four global climate strikes, the other strikes listed by FridaysForFuture (2020) are considered to be 'non-global strikes'. All strikes are predominantly linked to climate change, however, in the Global Climate Strike (3) (which lasted for one week) the topic biodiversity loss was more evident. The four global strikes took place with 2.3 M, 0.8 M, 7.3 and 1.2 M estimated participants in 137, 137, 183 and 166 countries, respectively (FridaysForFuture 2020). For activism, we focus especially on the first and the third global climate strikes in our study period.

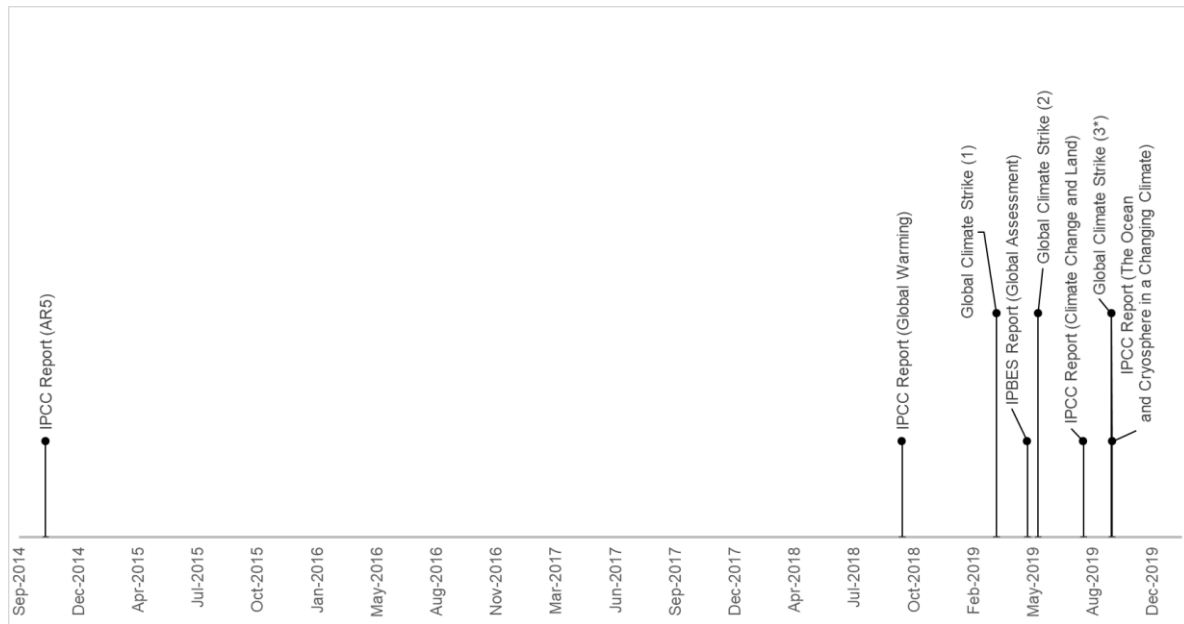


Figure 6: Overview of research-policy efforts and activism. * Indicates that the Global Climate Strike (3) lasted over a week. We selected in our analysis the first day of the strike as reference time.

7.4.3 Data Analysis

We identify national long-term (2011-2019) and medium-term (2016-2019) trends by using a linear regression model:

$$y_{scm} = \alpha + \rho t_{scm} + e_{scm} \quad (1)$$

where $y_{s,c,m}$ is the search volume of topic s (either climate change or biodiversity) in country c in month m . We run the model separately per topic and country. To account for heteroscedasticity and serial-correlation we use heteroscedasticity and serial-correlation consistent (HAC) standard errors in the estimations.

Furthermore, to identify regional trend changes after research-policy efforts or strikes we employ a by country population weighted linear regression model:

$$\Delta y_{scrt} = \alpha + \gamma Z_t + \beta X_c + e_{scrt} \quad (2)$$

where $\Delta y_{s,c,r,t}$ is the first difference of search volume of topic s in country c between period $t - 1$ and t : $(y_{r,t-1} - y_{r,t})$. r indicates the region of a country and t indicates either a week, w , (week-perspective) or a four week period, l , (four-week-perspective). For the weekly estimation we consider twelve weeks before and after the week in which the reports was released or the strike took place and four weekly estimation we consider six four-week periods before and after the four-week period in which the report was released or the strike took place. The week-perspective and four-week-perspective relate to each other as: $l_1 = w_1 + \dots + w_4, \dots, l_6 = w_{21} + \dots + w_{24}$ and $l_0 = w_0 +$

$\dots + w_{-3}, \dots, l_{-6} = w_{-23} + \dots + w_{-27}$. $Z_{r,t}$ indicates the week or the four-week period and X_r a set of country fixed effects. We run the model separately per topic and region. We correct standard errors by clustering over nations and time. Furthermore, to identify national and global correlations between research-policy efforts or strikes and interest level change in the time after the event, we use linear regression model:

$$y_{sct} = \alpha + \gamma Z_t + e_{sct} \quad (3)$$

For global correlations this model additionally includes country dummies (i.e. βX_c) and it is population weighted.

7.5 Data Availability

The data we downloaded (January 22 2020) from Google Trends will be available upon request (after publication).

7.6 Code Availability

All code to reproduce the data download and the analysis will be available upon request (after publication).

7.7 Acknowledgements

This study was supported by the Mercator Foundation Switzerland within a Zürich-Basel Plant Science Center PhD Fellowship program.

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7.9 Appendix

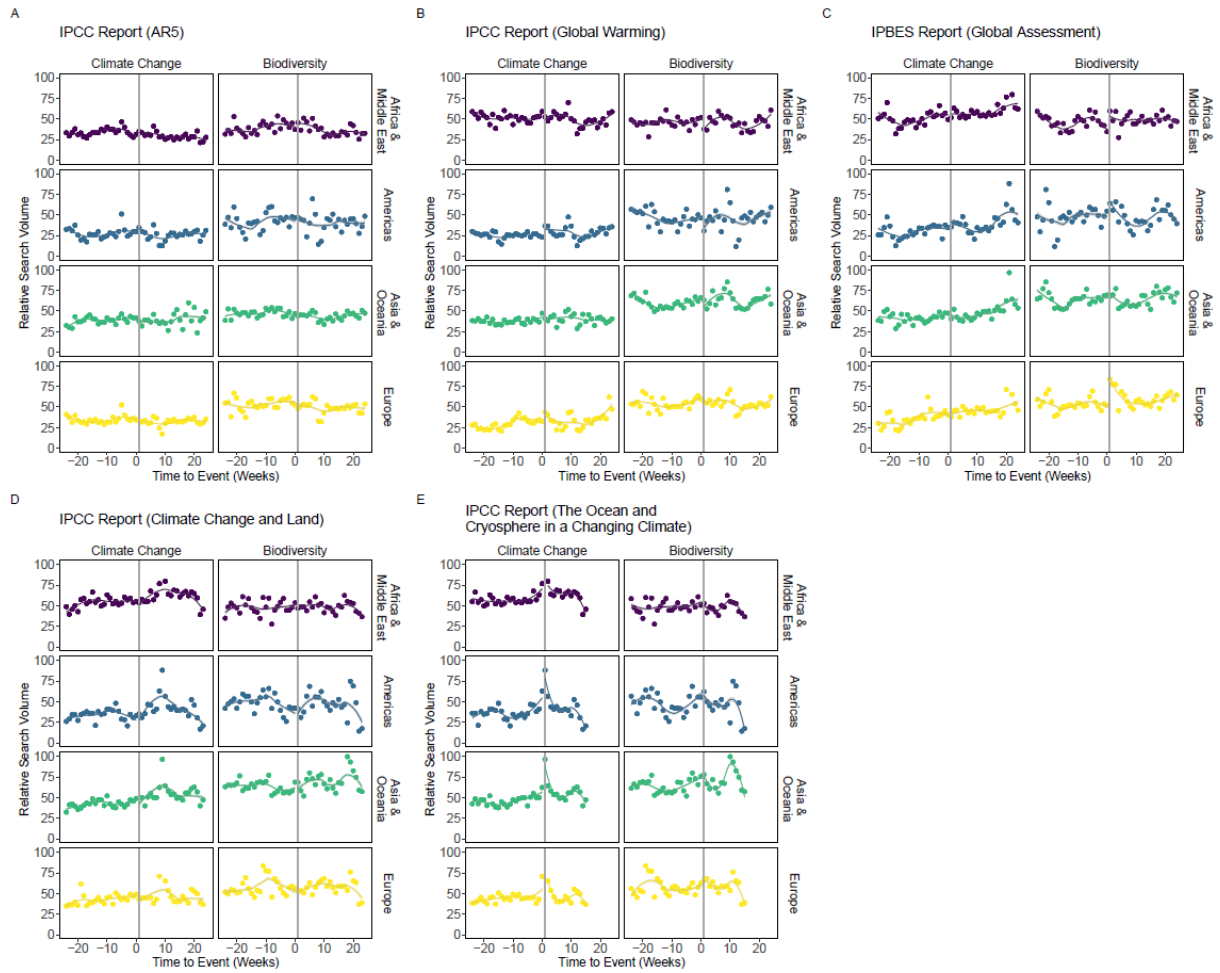


Figure A1: Climate change and biodiversity interest at the regional resolution from 24 weeks before to 24 four weeks after research-policy efforts in levels. Public interest is measured by the relative Google search volume.

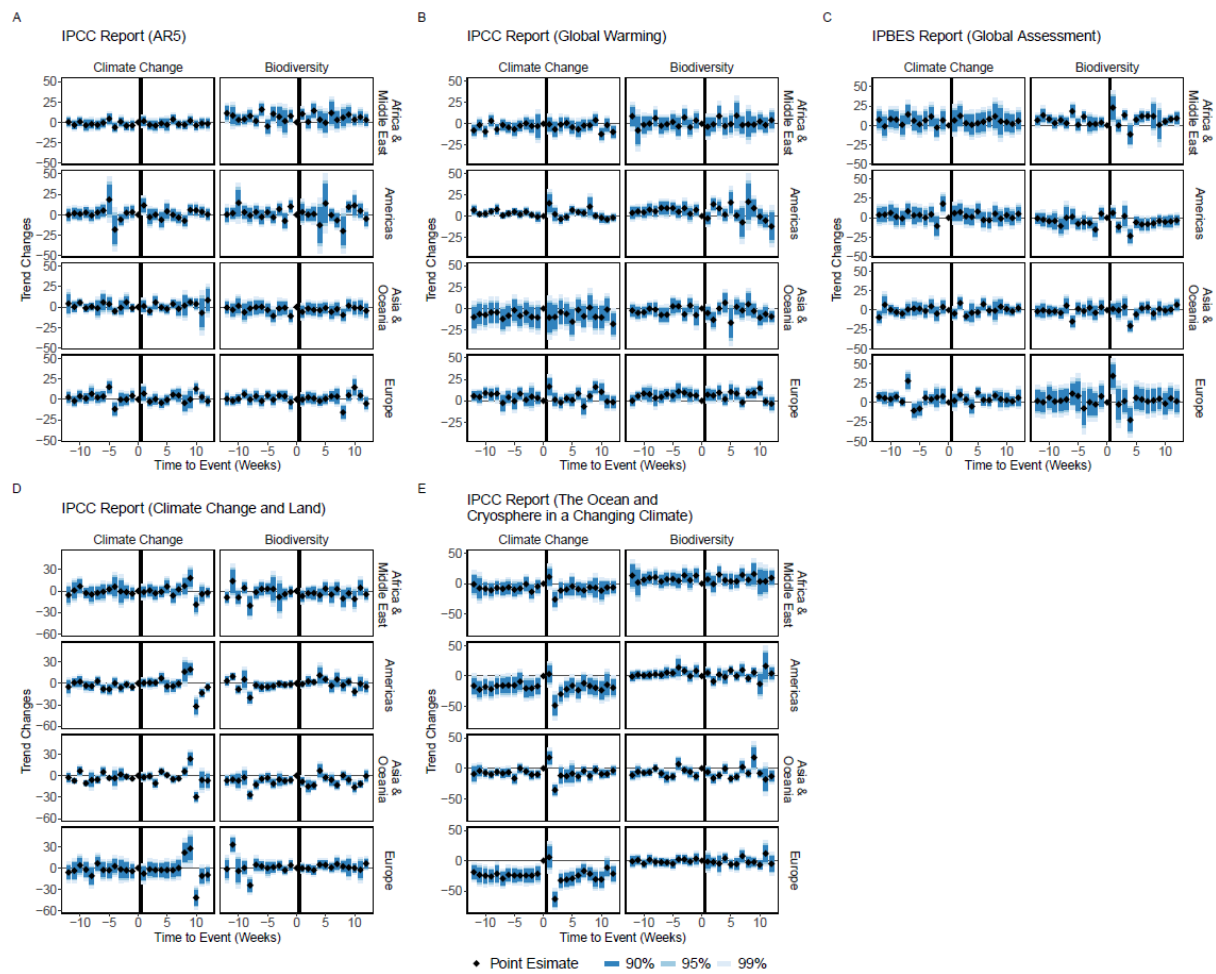


Figure A2: Weekly regional direct relationships of research-policy efforts on public climate change and biodiversity interest trends. Public interest is measured by the relative Google search volume. The x-axis shows 12 weeks before and after a research-policy effort. The diamonds are point estimates and the bars show the 90%, 95% and 99% confidence intervals. Point estimates and confidence intervals are from a population weighted linear regression model. We correct standard errors by clustering over nations and time.

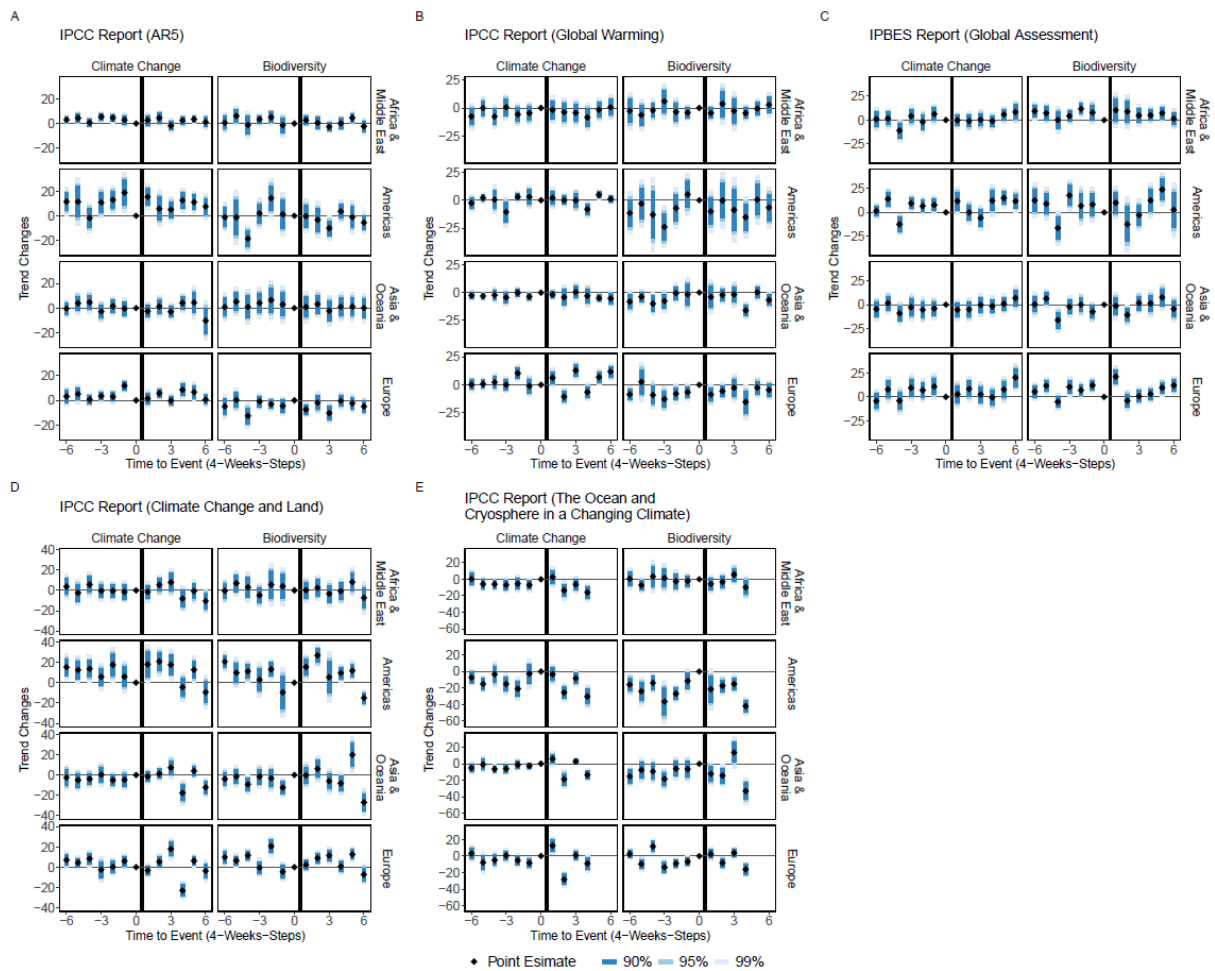


Figure A3: Four-weekly regional direct relationships of research-policy efforts on public climate change and biodiversity interest trends. Public interest is measured by the relative Google search volume. The x-axis shows 24 weeks before and after a research-policy effort in 4-week-steps. The diamonds are point estimates and the bars show the 90%, 95% and 99% confidence intervals. Point estimates and confidence intervals are from a population weighted linear regression model. We correct standard errors by clustering over nations and time.

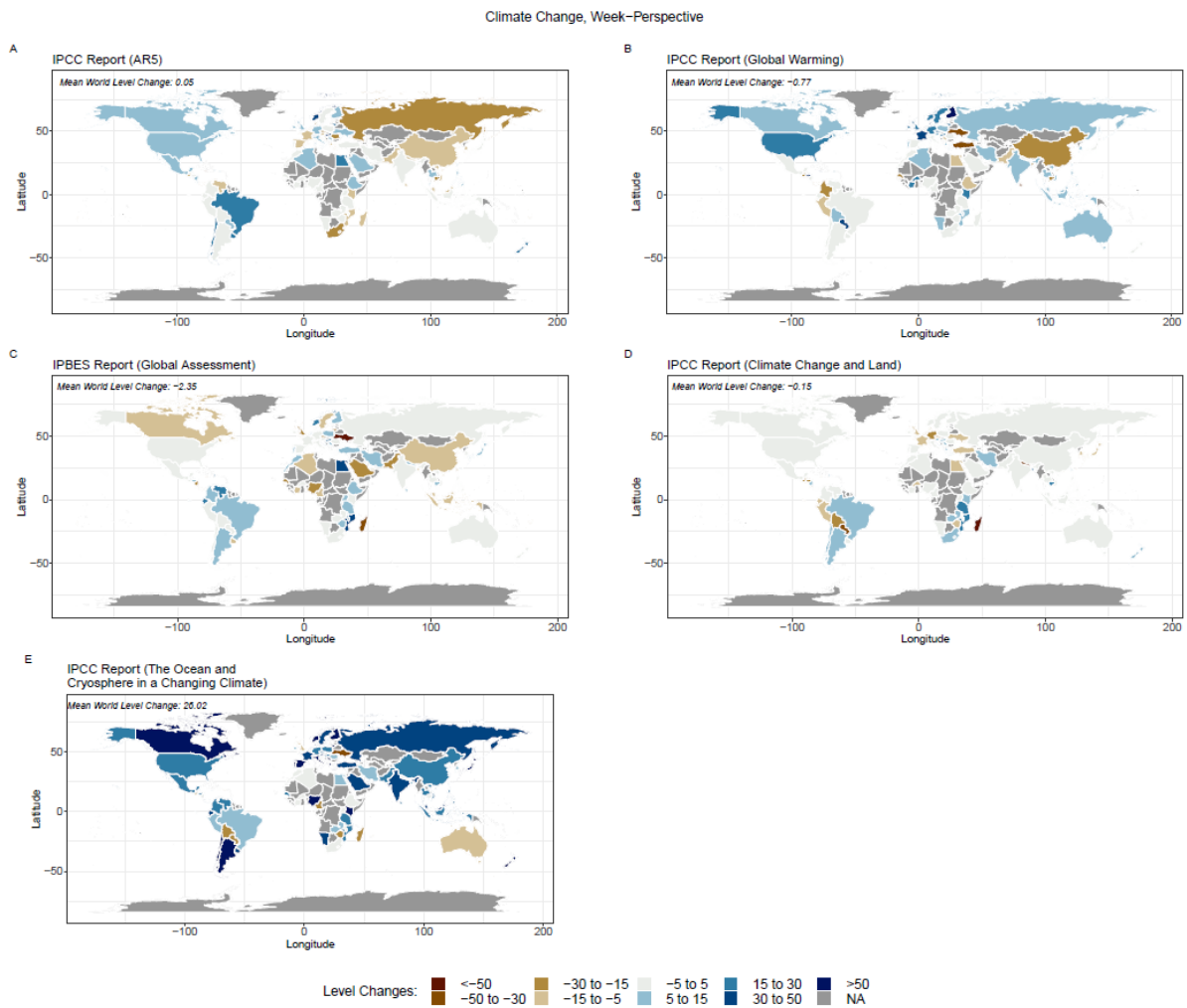


Figure A4: National interest level changes in climate change in the week after a research-policy effort compared to the week in which a research-policy effort took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population. Note that the week of the IPCC report (The Ocean and Cryosphere in a Changing Climate) release coincided with the Global Climate Strike (3).

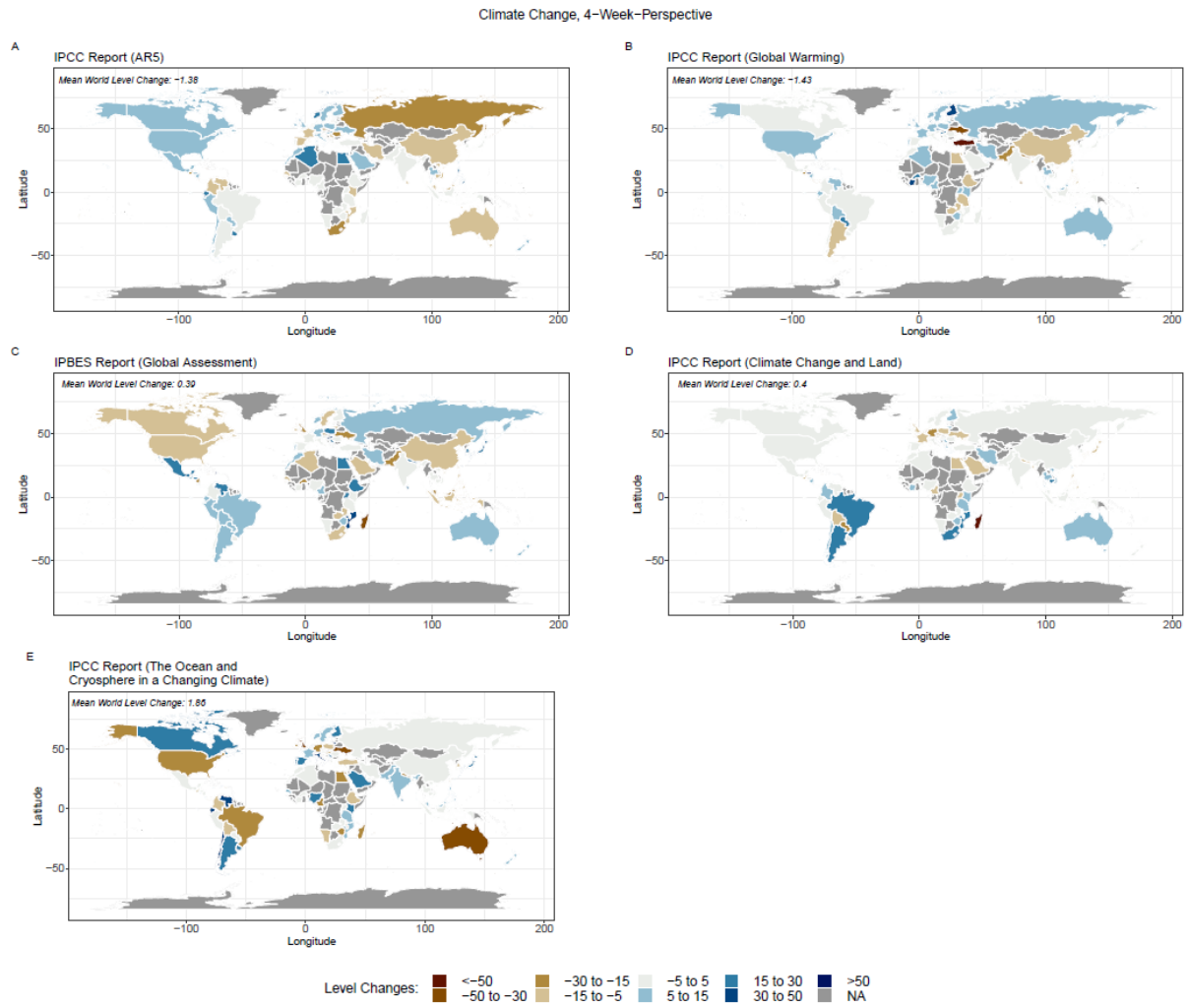


Figure A5: National interest level changes in climate change in the four-week period after a research-policy effort compared to the four-week period in which a research-policy effort took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

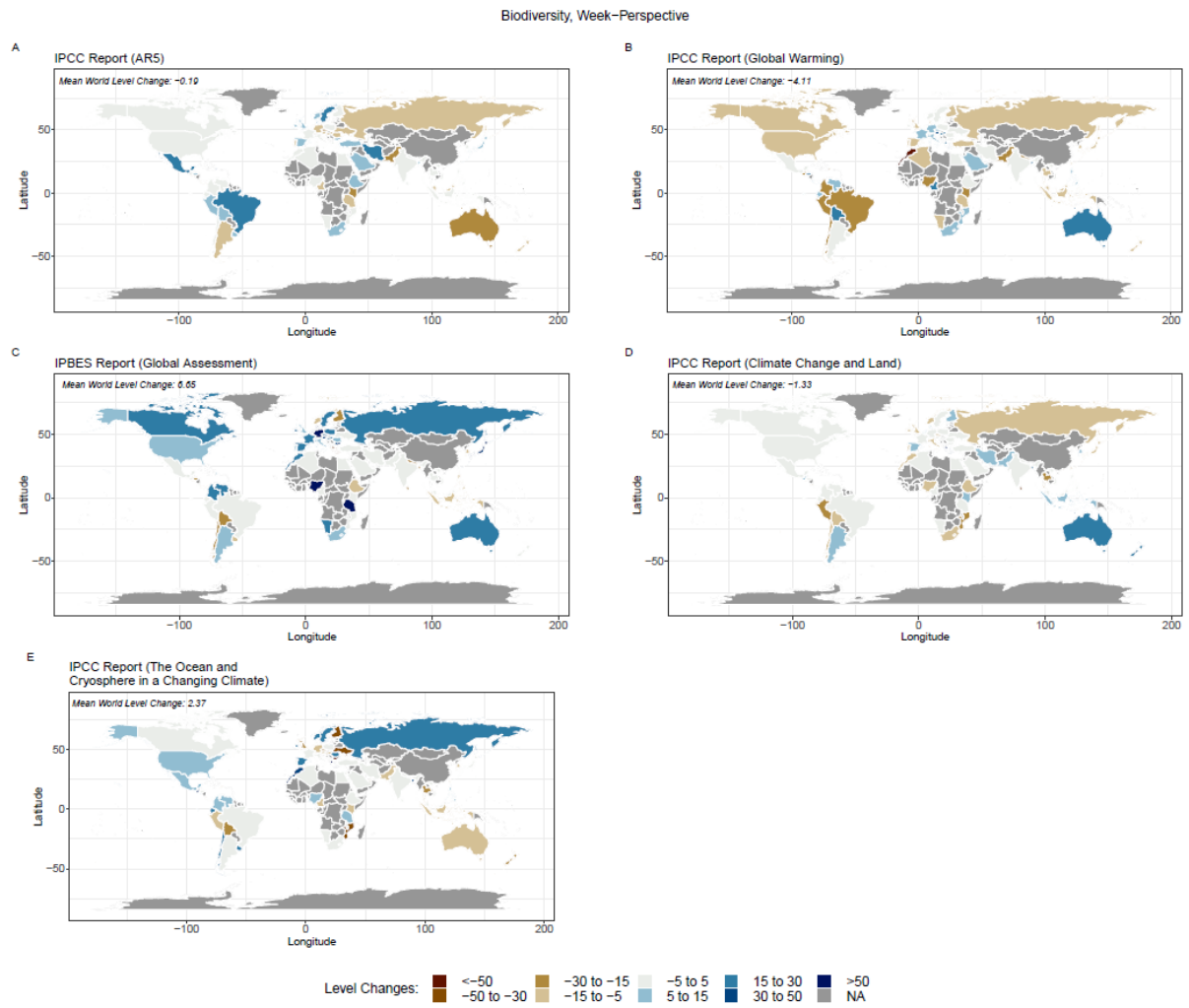


Figure A6: National interest level changes in biodiversity in the week after a research-policy effort compared to the week in which a research-policy effort took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

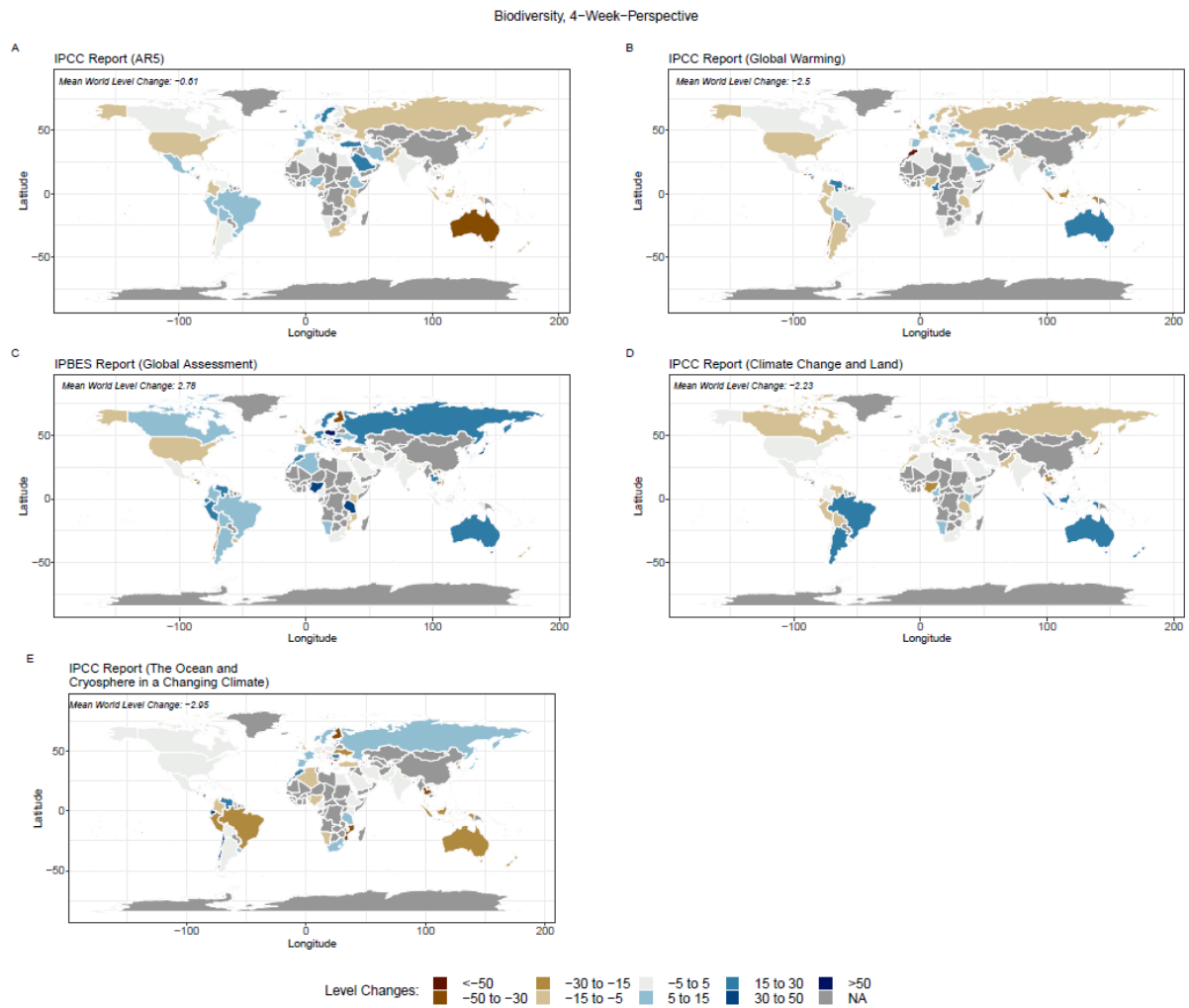


Figure A7: National interest level changes in biodiversity in the four-week period after a research-policy effort compared to the four-week period in which a research-policy effort took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

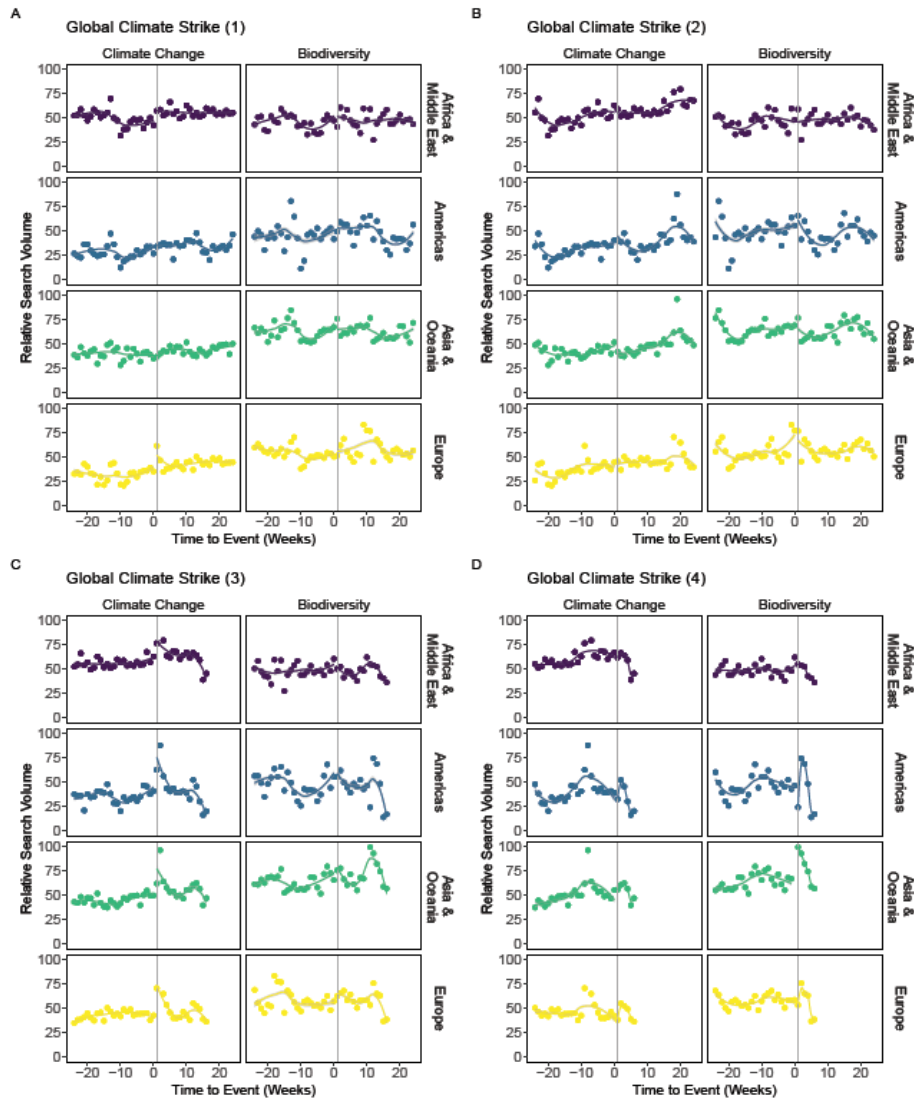


Figure A8: Climate change and biodiversity interest at the regional resolution from 24 weeks before to 24 four weeks after research-policy efforts in levels. Public interest is measured by the relative Google search volume.

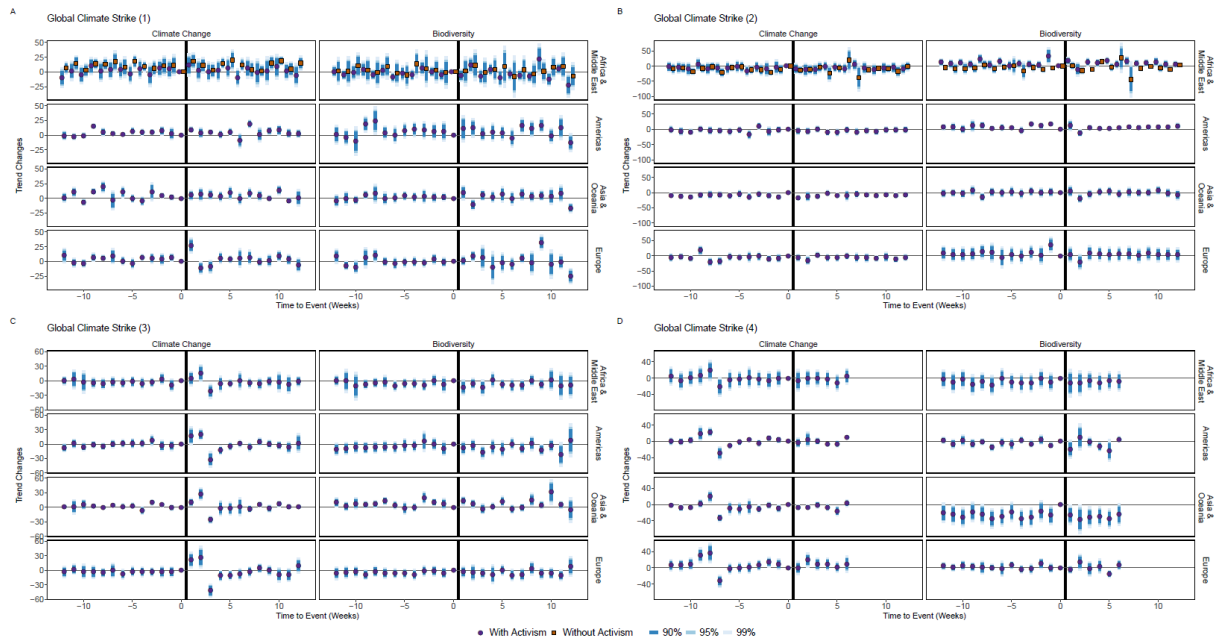


Figure A9: Weekly regional direct relationships of global strikes on public climate change and biodiversity interest trends. Public interest is measured by the relative Google search volume. The x-axis shows 12 weeks before and after a strike. Changes are indicated for countries with (purple cycles) and without (orange squares) strikes separately. Countries without strikes are only included if at least three countries in a region was without activism. The circles and squares are point estimates and the bars show the 90%, 95% and 99% confidence intervals. Point estimates and confidence intervals are from a population weighted linear regression model. We correct standard errors by clustering over nations and time.

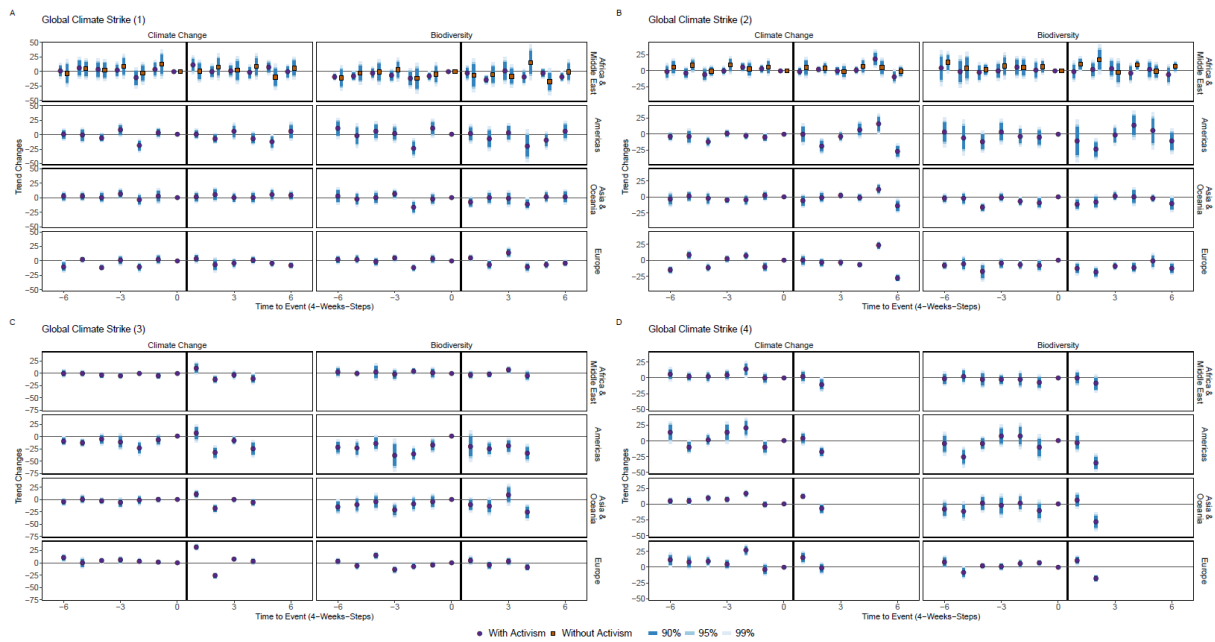


Figure A10: Four-weekly regional direct relationships of global strikes on public climate change and biodiversity interest trends. Public interest is measured by the relative Google search volume. The x-axis shows 24 weeks before and after a strike in 4-week-steps. Changes are indicated for countries with (purple circles) and without (orange squares) strikes separately. Countries without strikes are only included if at least three countries in a region was without activism. The circles and squares are point estimates and the bars show the 90%, 95% and 99% confidence intervals. Point estimates and confidence intervals are from a population weighted linear regression model. We correct standard errors by clustering over nations and time.

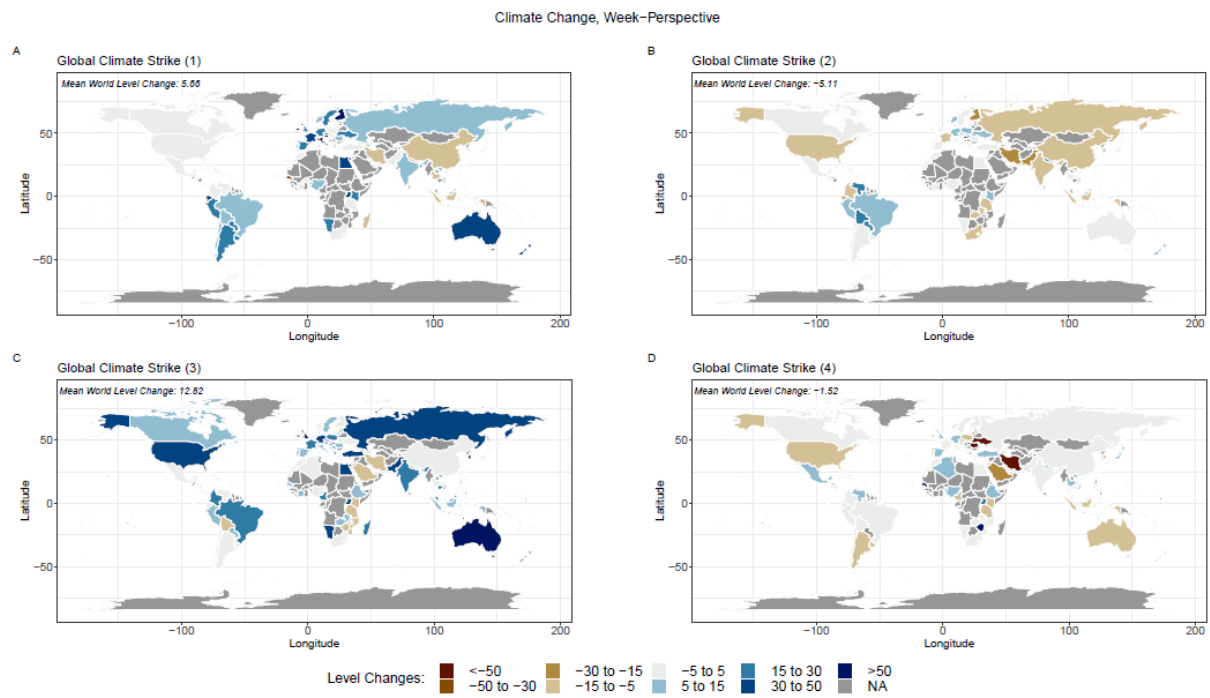


Figure A11: National interest level changes in climate change in the week after a strike compared to the week in which a strike took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

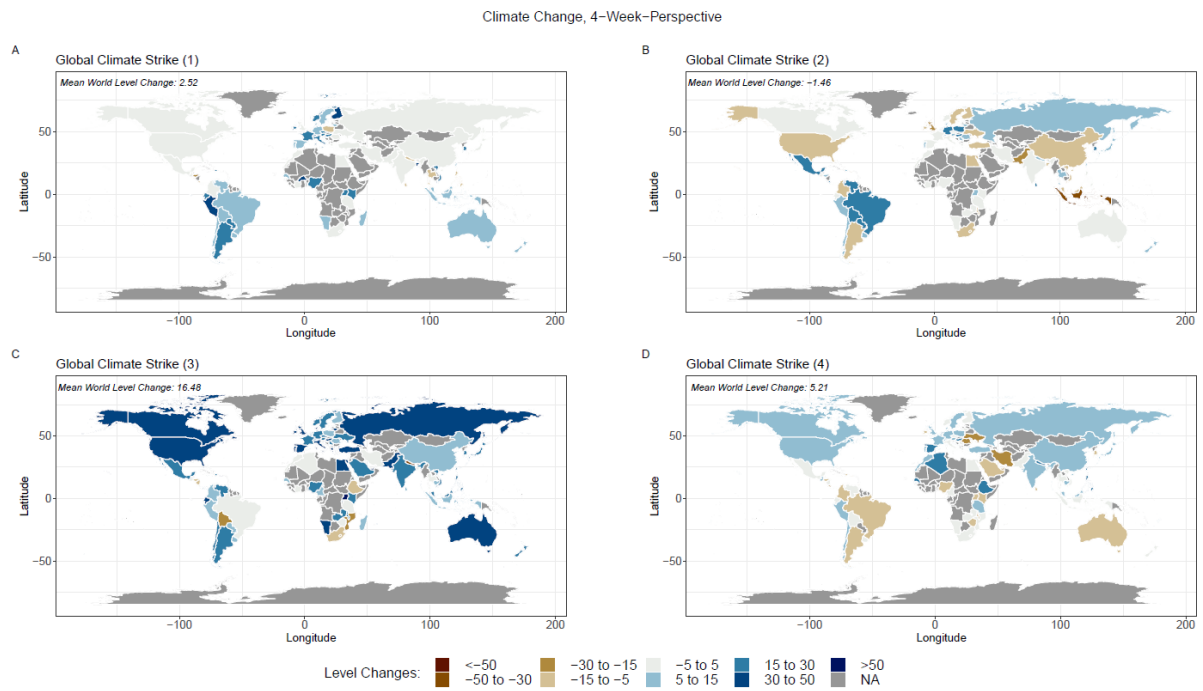


Figure A12: National interest level changes in climate change in the four-week period after a strike compared to the four-week period in which a strike took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

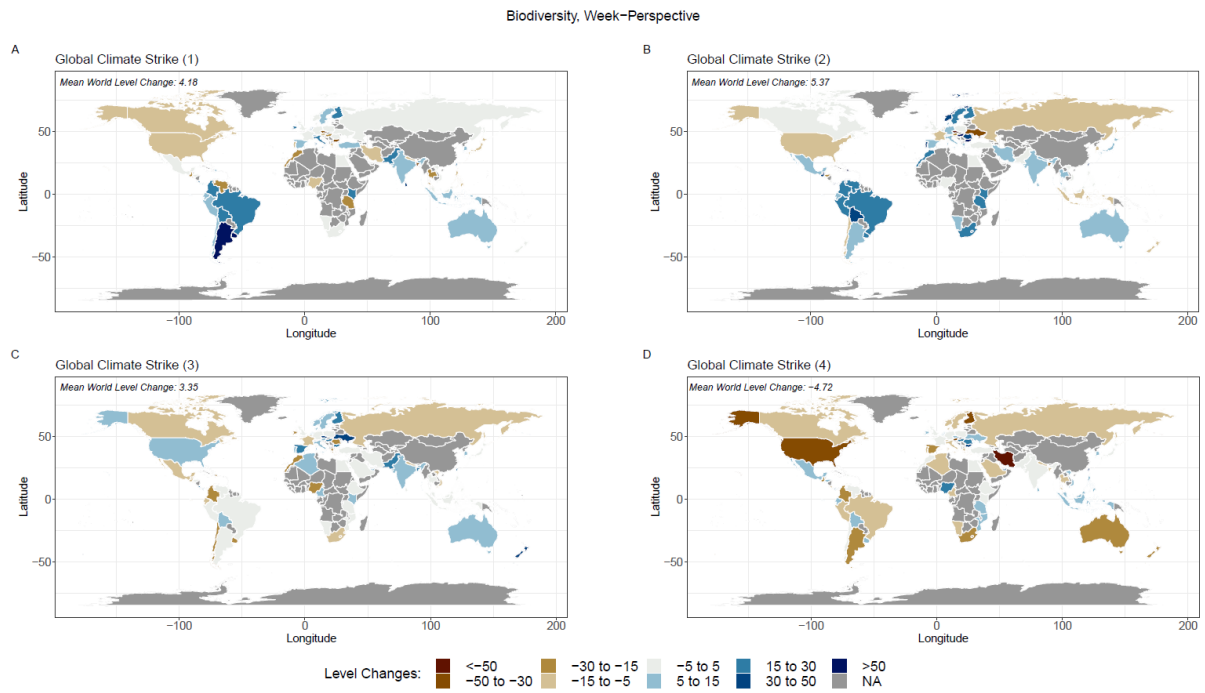


Figure A13: National interest level changes in biodiversity in the four-week period after a strike compared to the four-week period in which a strike took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

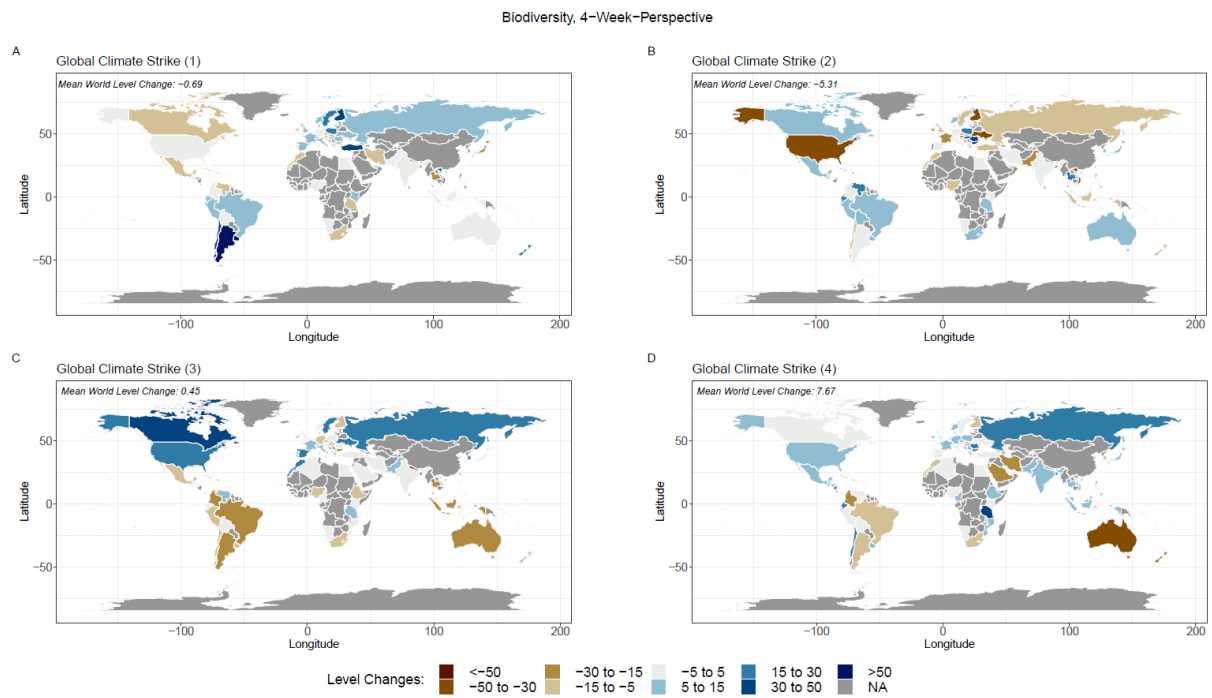


Figure A14: National interest level changes in biodiversity in the four-week period after a strike compared to the four-week period in which a strike took place. Public interest is measured by the relative Google search volume. The mean world level change is weighted by population.

Table A1: Monthly and weekly Google Trends data availability per country. Only countries are listed that have at least for one topic available data.

Country	Region	Monthly		Weekly	
		Climate Change	Biodiversity	Climate Change	Biodiversity
Afghanistan	Asia & Oceania	0.94	NA	0.55	NA
Albania	Europe	0.94	0.94	0.52	0.67
Algeria	Africa & Middle East	1	1	0.97	0.99
American Samoa	Asia & Oceania	0.37	NA	0.17	NA
Andorra	Europe	0.68	0.39	0.28	0.12
Angola	Africa & Middle East	0.99	0.99	0.46	0.71
Antigua and Barbuda	Americas	0.83	NA	0.27	NA
Argentina	Americas	1	1	1	1
Armenia	Asia & Oceania	0.93	0.85	0.5	0.37
Aruba	Americas	0.41	NA	0.21	NA
Australia	Asia & Oceania	1	1	1	1
Austria	Europe	1	1	1	0.98
Azerbaijan	Asia & Oceania	0.97	0.87	0.64	0.36
Bahamas	Americas	0.83	0.75	0.61	0.28
Bahrain	Africa & Middle East	0.94	0.72	0.72	0.35
Bangladesh	Asia & Oceania	1	1	1	0.99
Barbados	Americas	0.99	0.72	0.69	0.22
Belarus	Europe	0.99	0.92	0.81	0.53
Belgium	Europe	1	1	1	1
Belize	Americas	1	0.85	0.65	0.39
Benin	Africa & Middle East	1	0.96	0.89	0.62
Bermuda	Americas	0.7	NA	0.26	NA
Bhutan	Asia & Oceania	0.94	0.96	0.73	0.71
Bolivia	Americas	1	1	1	0.99
Bosnia and Herzegovina	Europe	1	0.96	0.8	0.51
Botswana	Africa & Middle East	0.99	0.99	0.8	0.53
Brazil	Americas	1	1	1	1
Brunei	Asia & Oceania	0.96	0.97	0.71	0.5
Bulgaria	Europe	1	1	0.84	0.93
Burkina Faso	Africa & Middle East	1	0.94	0.97	0.41
Burundi	Africa & Middle East	0.94	0.56	0.47	0.2
Cambodia	Asia & Oceania	1	1	0.96	0.72
Cameroon	Africa & Middle East	1	1	0.98	0.95
Canada	Americas	1	1	1	1
Cape Verde	Africa & Middle East	0.89	0.92	0.42	0.48
Cayman Islands	Americas	0.68	0.27	0.26	0.08
Chad	Africa & Middle East	0.75	0.58	0.21	0.2
Chile	Americas	1	1	1	1
China	Asia & Oceania	1	1	0.92	0.82
Colombia	Americas	1	1	1	1
Comoros	Africa & Middle East	0.69	0.34	0.21	0.11

Congo Republic	Africa & Middle East	0.96	0.77	0.56	0.29
Costa Rica	Americas	1	1	0.99	1
Ivory Coast	Africa & Middle East	1	1	0.94	0.8
Croatia	Europe	1	0.97	0.98	0.78
Cuba	Americas	1	0.89	0.82	0.44
Curacao	Americas	0.56	NA	0.21	NA
Cyprus	Africa & Middle East	1	0.99	0.88	0.72
Czech Republic	Europe	1	1	0.98	0.95
Denmark	Europe	1	1	1	0.99
Djibouti	Africa & Middle East	0.58	0.66	0.21	0.24
Dominica	Americas	0.8	0.23	0.32	0.1
Dominican Republic	Americas	1	1	1	0.98
East Timor	Asia & Oceania	0.89	0.8	0.36	0.32
Ecuador	Americas	1	1	1	1
Egypt	Africa & Middle East	1	1	0.99	0.92
El Salvador	Americas	1	1	0.97	0.92
Equatorial Guinea	Africa & Middle East	0.59	0.49	0.15	0.16
Estonia	Europe	0.99	0.96	0.82	0.72
Ethiopia	Africa & Middle East	1	1	1	0.99
Faroe Islands	Europe	0.27	NA	0.09	NA
Fiji	Asia & Oceania	1	0.94	0.97	0.68
Finland	Europe	1	1	1	0.97
France	Europe	1	1	1	1
French Guiana	Americas	NA	0.86	NA	0.42
French Polynesia	Asia & Oceania	NA	0.93	NA	0.34
Gabon	Africa & Middle East	0.85	0.85	0.51	0.35
Gambia	Africa & Middle East	0.9	NA	0.49	NA
Georgia	Asia & Oceania	0.96	0.94	0.63	0.65
Germany	Europe	1	1	1	1
Greece	Europe	1	1	1	0.96
Grenada	Americas	0.87	NA	0.35	NA
Guam	Asia & Oceania	0.93	0.7	0.51	0.18
Guatemala	Americas	1	1	0.99	0.96
Guernsey	Europe	0.7	NA	0.26	NA
Guinea	Africa & Middle East	0.93	0.9	0.51	0.36
Guyana	Americas	0.97	0.86	0.79	0.46
Haiti	Americas	1	0.97	0.75	0.67
Honduras	Americas	1	1	0.98	0.96
Hong Kong	Asia & Oceania	1	1	1	1
Hungary	Europe	1	1	0.99	0.92
Iceland	Europe	1	0.72	0.65	0.32
India	Asia & Oceania	1	1	1	1
Indonesia	Asia & Oceania	1	1	1	1
Iran	Africa & Middle East	1	1	1	1
Iraq	Africa & Middle East	1	0.92	0.79	0.64
Ireland	Europe	1	1	1	1
Isle of Man	Europe	0.77	NA	0.32	NA

Israel	Africa & Middle East	1	1	0.95	0.93
Italy	Europe	1	1	1	1
Jamaica	Americas	1	0.99	0.98	0.77
Japan	Asia & Oceania	1	1	1	1
Jersey	Europe	0.73	0.58	0.31	0.19
Jordan	Africa & Middle East	1	0.96	0.9	0.75
Kazakhstan	Asia & Oceania	1	0.96	0.9	0.78
Kenya	Africa & Middle East	1	1	1	1
Kiribati	Asia & Oceania	0.35	NA	0.1	NA
Korea (South)	Asia & Oceania	1	1	1	1
Kosovo	Europe	0.97	0.85	0.6	0.41
Kuwait	Africa & Middle East	1	0.94	0.85	0.57
Kyrgyzstan	Africa & Middle East	0.97	0.77	0.62	0.37
Laos	Asia & Oceania	0.99	0.99	0.63	0.46
Latvia	Europe	1	0.94	0.83	0.58
Lebanon	Africa & Middle East	1	1	0.91	0.74
Lesotho	Africa & Middle East	0.99	0.7	0.73	0.27
Liberia	Africa & Middle East	0.9	0.75	0.59	0.37
Libya	Africa & Middle East	0.72	0.76	0.4	0.33
Liechtenstein	Europe	0.41	NA	0.14	NA
Lithuania	Europe	1	0.97	0.91	0.53
Luxembourg	Europe	1	0.89	0.73	0.54
Macao	Asia & Oceania	0.73	0.7	0.44	0.28
North Macedonia	Europe	0.97	0.87	0.64	0.41
Madagascar	Africa & Middle East	1	1	0.93	0.87
Malawi	Africa & Middle East	1	0.97	0.96	0.52
Malaysia	Asia & Oceania	1	1	1	1
Maldives	Asia & Oceania	0.97	0.85	0.61	0.34
Mali	Africa & Middle East	0.99	0.96	0.84	0.49
Malta	Europe	1	0.9	0.79	0.5
Marshall Islands	Asia & Oceania	0.89	NA	0.36	NA
Mauritania	Africa & Middle East	0.77	0.65	0.31	0.23
Mauritius	Africa & Middle East	1	0.96	0.92	0.74
Mexico	Americas	1	1	1	1
Micronesia	Asia & Oceania	0.75	NA	0.34	NA
Moldova	Europe	0.97	0.86	0.5	0.58
Mongolia	Asia & Oceania	0.97	0.77	0.65	0.28
Montenegro	Europe	0.93	0.72	0.41	0.28
Morocco	Africa & Middle East	1	1	0.98	0.96
Mozambique	Africa & Middle East	1	1	0.92	0.92
Myanmar	Asia & Oceania	1	0.94	0.86	0.7
Namibia	Africa & Middle East	1	1	1	1
Nepal	Asia & Oceania	1	1	1	0.99
Netherlands	Europe	1	1	1	1
New Caledonia	Asia & Oceania	0.8	0.83	0.22	0.42
New Zealand	Asia & Oceania	1	1	1	1
Nicaragua	Americas	1	1	0.98	0.89

Niger	Africa & Middle East	1	0.77	0.73	0.3
Nigeria	Africa & Middle East	1	1	1	1
Northern Mariana Islands	Asia & Oceania	0.62	0.44	0.23	0.11
Norway	Europe	1	1	1	0.99
Oman	Africa & Middle East	1	0.99	0.78	0.69
Pakistan	Asia & Oceania	1	1	1	1
Panama	Americas	1	1	0.98	0.99
Papua New Guinea	Asia & Oceania	1	0.86	0.84	0.55
Paraguay	Americas	1	1	0.93	0.84
Peru	Americas	1	1	1	1
Philippines	Asia & Oceania	1	1	1	1
Poland	Europe	1	1	1	1
Portugal	Europe	1	1	1	1
Puerto Rico	Americas	1	0.99	0.91	0.81
Qatar	Africa & Middle East	1	0.96	0.85	0.63
Romania	Europe	1	1	0.99	0.99
Russian Federation	Europe	1	1	1	0.99
Rwanda	Africa & Middle East	1	0.97	0.91	0.65
Saint Kitts and Nevis	Americas	0.46	NA	0.18	NA
Saint Lucia	Americas	0.9	NA	0.49	NA
Saint Vincent and the Grenadines	Americas	0.83	0.46	0.41	0.13
Samoa	Asia & Oceania	0.97	0.46	0.56	0.1
San Marino	Europe	0.15	NA	0.03	NA
Sao Tome and Principe	Africa & Middle East	0.49	0.38	0.11	0.13
Saudi Arabia	Africa & Middle East	1	1	0.99	0.93
Senegal	Africa & Middle East	1	0.99	0.94	0.68
Serbia	Europe	1	1	0.97	0.88
Seychelles	Africa & Middle East	0.75	NA	0.28	NA
Sierra Leone	Africa & Middle East	0.9	0.75	0.58	0.22
Singapore	Asia & Oceania	1	1	1	1
Slovakia	Europe	1	1	0.88	0.82
Slovenia	Europe	1	1	0.84	0.81
Solomon Islands	Asia & Oceania	0.99	0.65	0.68	0.21
Somalia	Africa & Middle East	0.93	0.66	0.53	0.25
South Africa	Africa & Middle East	1	1	1	1
South Sudan	Africa & Middle East	0.79	NA	0.37	NA
Spain	Europe	1	1	1	1
Sri Lanka	Asia & Oceania	1	1	0.98	0.99
Sudan	Africa & Middle East	1	0.94	0.86	0.52
Suriname	Americas	0.89	0.76	0.43	0.28
Swaziland	Africa & Middle East	0.99	0.44	0.52	0.28
Sweden	Europe	1	1	1	0.99
Switzerland	Europe	1	1	1	1
Syria	Africa & Middle East	0.87	0.73	0.41	0.34
Taiwan	Asia & Oceania	1	1	1	1

Tajikistan	Asia & Oceania	0.85	NA	0.32	NA
Tanzania	Africa & Middle East	1	1	0.99	0.95
Thailand	Asia & Oceania	1	1	1	1
Togo	Africa & Middle East	0.99	0.79	0.71	0.35
Tonga	Asia & Oceania	0.79	NA	0.34	NA
Trinidad and Tobago	Americas	1	1	0.92	0.8
Tunisia	Africa & Middle East	1	1	0.9	0.9
Turkey	Africa & Middle East	1	1	1	1
Uganda	Africa & Middle East	1	1	0.99	0.88
Ukraine	Europe	1	1	0.98	0.97
United Arab Emirates	Africa & Middle East	1	1	1	0.99
United Kingdom	Europe	1	1	1	1
United States	Americas	1	1	1	1
Uruguay	Americas	1	1	0.98	0.95
Uzbekistan	Asia & Oceania	0.96	0.61	0.46	0.27
Vanuatu	Asia & Oceania	0.97	NA	0.65	NA
Venezuela	Americas	1	1	1	1
Viet Nam	Asia & Oceania	1	1	1	1
Virgin Islands (US)	Americas	0.52	NA	0.2	NA
Yemen	Africa & Middle East	0.79	0.42	0.27	0.12
Zambia	Africa & Middle East	1	1	0.99	0.84
Zimbabwe	Africa & Middle East	1	1	1	0.86

8 Thesis Appendix 3 – Tracking Societal Concerns on Pesticide – A Google Trends Analysis

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Abstract

Pesticide use and its environmental impact generate polarized societal discussions. This has important impacts for the political economy of pesticide policies. Against the background of two upcoming popular initiatives on pesticides in Switzerland, we analyze public interest in pesticides and plant protection products over time utilizing Google Trends data from 2011 till 2019. We make use of that the term ‘pesticides’ has a more negative connotation compared to ‘plant protection products’. We find that the aggregated public interest in pesticides and plant protection products substantially increased over the last years, especially since 2017. This trend can be attributed to search terms related to pesticides while the interest in search terms related to plant protection products remained stable. We find evidence, which supports the hypothesis that increasing concerns on pesticide use contributed to the launches of the popular initiatives. Moreover, our results support also that the popular initiative launches amplified these concerns. We conclude that Google Trends is a useful tool to uncover ongoing discussions early on, which otherwise might be unobserved. Therefore, it can generate useful insights and contribute to agricultural policy problem framing.

Keywords: google trends, pesticide, agricultural policy, popular initiative, public interest

8.1 Introduction

The agricultural sector and agricultural policy are facing high societal pressure from different actors, ranging from industry to NGOs to consumers. This affects agricultural policy as societal preferences matter for agricultural policy development (e.g. Buckwell et al. 2017). Policy design and communication need to account for public's perception to ensure policy acceptance. New information and communication technologies change public's access to information about agriculture and agricultural policies. This may also increase relevance of environmental problems in the context of agriculture, which traditionally had been of less importance in agricultural policy problem framing.

Google Trends has emerged a powerful tool to quantify public interest in a topic. Trends are reflected in the Google search volume of specific terms (e.g. Anderegg and Goldsmith 2014). Thus, Google Trends data has been suggested to complement other data for policy analysis (e.g. Blume et al 2014) and it had been used to analyze people's preferences (e.g. interest in climate change or conservation) and behavior (e.g. search inquiries as measure for unemployment measures or influenza outbreaks) as well as for price prediction (e.g. stock or agricultural commodity prices; Da et al 2011, Anderegg and Goldsmith 2014, Nghiem et al 2016, Mišečka et al. 2019). However, little empirical knowledge exists on the development of public interest and perception in environmental externalities from agriculture. We here contribute to close this gap and use Google Trends data to analyze the Swiss case of public interest in pesticide over time. In our analysis, we include national and international political activities and reports on pesticides. We identify structural breaks reflecting discontinuities in the mean and/or slope of the public interest. Our analysis exploits recent policy discussions on pesticides (or plant protection products) use and two recently initiated popular initiatives in that field in Switzerland. Forms of direct democracy, such as popular initiatives or referendums, are still today important political instruments in many countries (Qvortrup 2014, Qvortrup 2017, Huber and Finger 2019). Additionally, we make use of that the term 'pesticides' has a negative connotation compared to 'plant protection products'. In this context, we address two questions: a) how societal concerns on pesticides emerge over time and b) how public discussions change the perception of agricultural practices. The use of Google Trends data to inform policy problem framing in agriculture provides several advantages. First, it is not based on surveys and thus freely accessible and unbiased data that is readily available. Second, it provides long term perspective covering coherent data collection over many years.

We find that the interest in pesticides and plant protection products increased considerably over time, which increase was driven by an increased interest in pesticides. Moreover, our analysis shows structural breaks in these interest around the launches of the popular initiatives.

The remainder of this paper is structured as follows. In the next section we present the methods and the data sources of this paper as well as background about political activities and report releases associated with pesticide use. This is followed by a result section and finally by a discussion and conclusion section.

8.2 Methods & Data

8.2.1 Estimation of Structural Breaks

We estimate a linear model of search volume as function of time with m structural breaks ($m + 1$ regimes; Bai and Perron 2003)⁶³:

$$y_t = x_t' \beta_j + u_t \quad t = T_{j-1} + 1, \dots, T_j \quad (1)$$

for $j = 1, \dots, m + 1$. y_t is the search volume at time t , x_t' is a $(k \times 1)$ vector of dependent variables, including the intercept and time, and u_t is the disturbance at time t . β_j and T_1, \dots, T_m are the unknown coefficients and structural breaks, respectively, and are jointly estimated. Furthermore, we select the number of structural breaks based on the Bayesian information criterion.

8.2.2 Google Trends Data

Google, which hosts the most widely used search engine (Sterling 2020), provides relative trend data in a period for search terms⁶⁴. We downloaded monthly Google Trends data between January 2011 and December 2019 for search terms related to pesticides and plant protections and the two initiatives in Switzerland using the R package 'gtrendsR' (Massicotte and Eddelbuettel 2019). The main search term group, 'pesticides and plant protections', was divided in two subgroups, 'pesticides' and 'plant protection products'. We used search terms in the main official languages in Switzerland (i.e. German, French and Italian) for download request from Google, however, only German and French search terms yielded in available data (Table A1). Next, we computed for each main group (i.e. 'pesticides and plant protection products' and 'initiatives'), subgroups (i.e. 'pesticides' or 'plant protection products') and search terms⁶⁵ the search volume (Appendix 2).

Google provides random samples of all search inquires (Google 2020). This can lead to sampling noise, especially for small samples or for less popular search terms. We addressed this concern by downloading data on 33 days and following the procedure proposed by Carrière-Swallow and Labbé

⁶³ We use the R package 'strucchange' (Zeileis et al 2002, Zeileis et al 2003) for the estimation.

⁶⁴ The Google Search Volume Index ranges between 0 and 100 and the data is normalized and rescaled (Google 2020). We note that Google Trends penalizes search terms with low search volume. This has two implications for us. First, some of the search terms resulted in no data (Table A1). Second, in some months with low search volume Google returns zeros even when the term had some search inquiries (Da et al. 2011).

⁶⁵ We rescaled search volume of the individual search terms as Google only allows downloading up to five search terms at a time (Appendix 2).

(2013) (see Appendix 3 for details). Using this procedure we show that generally sampling noise is no, or a minor, concern in our case (Fig. A1).

8.2.3 Political Activities & Report Releases

We focus on political activities and national reports related to the popular initiatives and pesticides (Table A2). The political activities include events related to start of collecting signatures, submission of initiative and the publication of the official response by the Swiss Federal Council. National reports include two important publications by the Swiss Federal Institute of Aquatic Science and Technology and one by the Swiss Association of Cantonal Chemists. Additionally, we consider two important report releases by United Nations units⁶⁶ on glyphosate. Interesting in this context is that the earlier report includes a 'negative' and the later report a 'positive' assessment of glyphosate⁶⁷.

8.3 Results

We find that public interest in pesticides and plant production products clearly increased between 2011 and 2019 (Fig. 1, Panel A). The search volume of pesticides and plant production products doubled from 2011 to 2019. The increase over time is characterized by two structural breaks in the beginning of 2014 and 2017. The first structural break occurred about three years before the launch of the two initiatives. The second structural break occurred at about the time of their launch.

The interest in the topic of pesticides and plant production products peaked after political activities related to the popular initiatives. Yet, these political activities often concur with national report releases on pesticides in general and pesticide residuals in the Swiss drinking water. The interest in the popular initiatives constantly increased after they were launched. After the first and the second international report releases, the earlier reporting a 'negative' and the later a 'positive' assessment of glyphosate, we find an increased and a decreased interest in the pesticide topic, respectively. However, these changes were rather moderate.

⁶⁶ The United Nations units are either the World Health Organization (WHO) or the WHO and the Food and Agriculture Organisation (FAO).

⁶⁷ We classify the earlier report as 'negative' as it reports "Glyphosate is probably carcinogenic to humans ..." (WHO and FAO 2017) and the later report as 'positive' as it reports "...glyphosate is unlikely to pose a carcinogenic risk to humans from exposure through the diet." (WHO 2016).

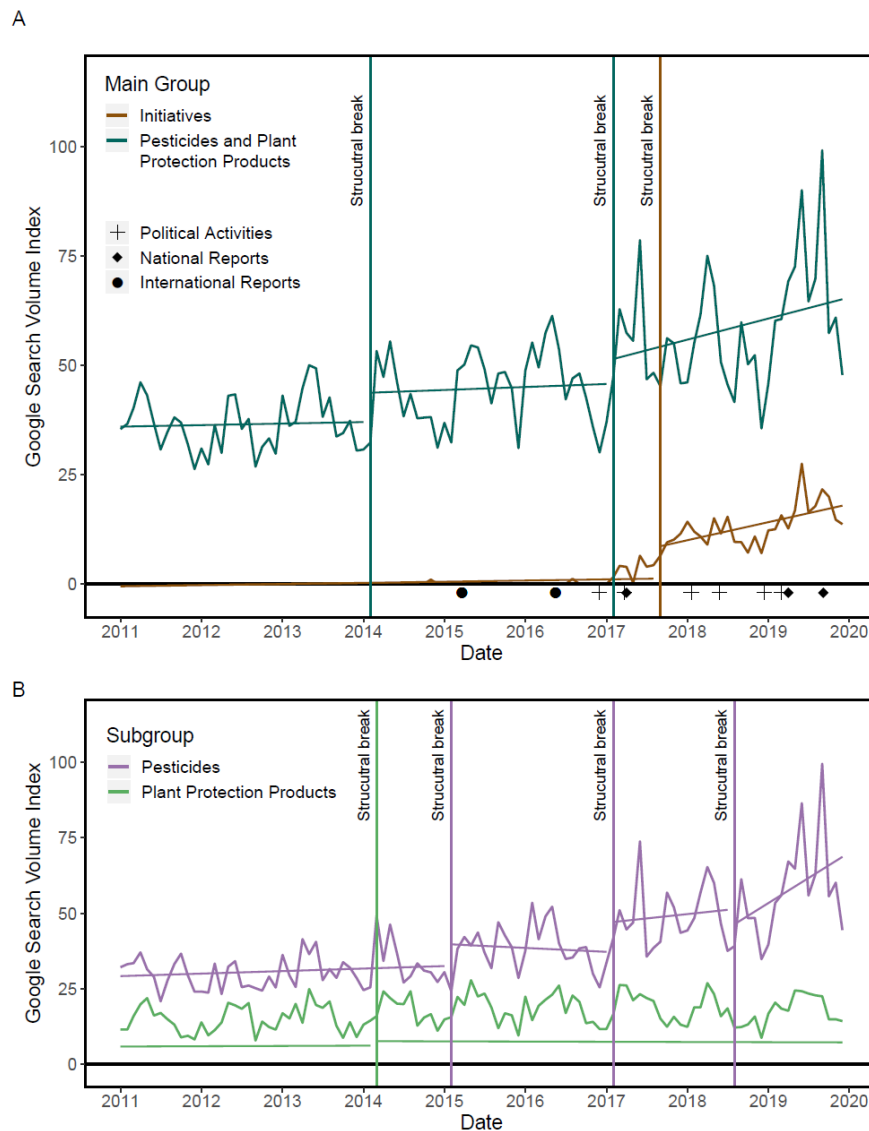


Figure 1: Interest over time for search terms related to pesticides and the popular initiatives in Switzerland. Panel A shows interest over time in the two main groups (pesticides or plant protection products or initiatives) and important related events. Shapes indicate the happening of an event. Panel B shows interest over time in the subgroups (pesticide or plant protection products).

Interestingly, the increase in public interest differs between the search terms related to pesticides and to plant protection products. While the interest in pesticides strongly increased over time the interest in plant protection products remained rather stable (Fig. 1, Panel B). This indicates that changes in the aggregated search volume of these search terms is generally linked to changes in the search volume of search terms related to pesticides. In contrast to the overall results, we find three structural breaks for interest in pesticides in our data analysis. The second structural break of interest in pesticides was identical to the second break of the aggregated interest in pesticides and plant production products. This break interestingly happened at the time of the initiative launches and just before interest in the

initiatives started to increase. The third structural breaks of the subgroup pesticides occurred about one and a half year after the initiative launches.

8.4 Discussion & Conclusion

We find that the overall interest (expressed in search volume) in the topic of pesticides and plant protection products substantially increased between 2011 and 2019 in Switzerland. This trend was driven by an increased interest in pesticides, while the interest for plant protection products was rather stable over time. Moreover, our analysis shows that the search volume for pesticides started increasing at a higher rate about the time of the launches of the popular initiatives. This rate again increased about one and a half year after their launches.

The here identified trends for Switzerland are in contrast to those observed in most neighboring countries of Switzerland, i.e. Austria, Germany and Italy (Fig. A2). In these three countries, interest in pesticides and plant protection products were rather constant and interest between pesticides and plant production products did generally not diverge over time. France, as an exception of the neighboring countries, showed similar trends as Switzerland. Indeed, France, as Switzerland, faces high societal and political discussion on pesticides (e.g. Stokstad 2018).

Our findings suggest that, on average, an increasingly negative connotation is given to plant protection. Additionally, our results might also signal that different persons are conducting the search inquires (e.g. farmers vs. general public) and an increasing public interest in the pesticide use, its environmental impacts and pesticide policies. Furthermore, our findings provide evidence that support the hypothesis that increasing concerns about pesticide use contributed to the launching of the popular initiative. And, that the popular initiative launches amplified these concerns. By using Google Trends data we analyzed a large share of the population; but the results are restricted to the population that uses Google as tool to acquire information.

Our findings have two important policy implications. Firstly, a clear communication highlighting both negative but also positive aspects of plant protection might help to avoid strong perception biases of the public towards plant protection. Secondly, Google Trends data is a useful tool to uncover ongoing discussions early on, which are otherwise unobserved, and to complement traditional data sources. By using Google Trends data, authorities and stakeholders can react early on to communicate their viewpoints. Thus, Google Trends data can be very useful for setting the agenda of policy processes.

Our analysis of societal pesticide concerns and public initiatives in Switzerland also highlights important areas for future research, including: The investigation of societal pesticide concerns over time, between countries and what explains national and temporal differences. The analysis of societal

concerns of other (agricultural, environmental, etc.) topics connected to popular initiatives (or other forms of direct democracy) on the national and local level, in Switzerland and in other countries.

8.5 Data Availability

The data we downloaded from Google Trends will be available upon request (after publication).

8.6 Code Availability

All code to reproduce the data download and the analysis will be available upon request (after publication).

8.7 Acknowledgements

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8.9 Appendix 1

Table A1: List of search terms.

Group	Subgroup	Search term	Language	Data available
Pesticides	Plant protection products	Pflanzenschutzmittel	German	Yes
Pesticides	Plant protection products	Produit phytosanitaire	French	Yes
Pesticides	Plant protection products	Prodotto fitosanitario	Italian	No
Pesticides	Pesticides	Pestizid	German	Yes
Pesticides	Pesticides	Pestizide	German	Yes
Pesticides	Pesticides	Pesticide*	French	Yes
Pesticides	Pesticides	Pesticida	Italian	No
Initiatives	Clean Drinking Water and Healthy Food	Für sauberes Trinkwasser und gesunde Nahrung	German	No
Initiatives	Clean Drinking Water and Healthy Food	Trinkwasserinitiative	German	Yes
Initiatives	Clean Drinking Water and Healthy Food	Trinkwasser initiative	German	Yes
Initiatives	Clean Drinking Water and Healthy Food	Trinkwasser-initiative	German	No
Initiatives	Save Switzerland from synthetic pesticides	Für eine Schweiz ohne synthetische Pestizide	German	Yes
Initiatives	Save Switzerland from synthetic pesticides	Pestizid initiative	German	Yes
Initiatives	Clean Drinking Water and Healthy Food	Pour une eau potable propre et une alimentation saine	French	No
Initiatives	Clean Drinking Water and Healthy Food	Initiative pour l'eau potable	French	No
Initiatives	Clean Drinking Water and Healthy Food	Initiative eau propre	French	No
Initiatives	Save Switzerland from synthetic pesticides	Pour une Suisse libre de pesticides de synthese	French	No
Initiatives	Save Switzerland from synthetic pesticides	Pour une Suisse sans pesticides de synthese	French	No
Initiatives	Save Switzerland from synthetic pesticides	Initiative sur les pesticides de synthèse	French	No
Initiatives	Save Switzerland from synthetic pesticides	Zero pesticide	French	No
Initiatives	Save Switzerland from synthetic pesticides	Initiative pesticide	French	No
Initiatives	Clean Drinking Water and Healthy Food	Acqua potabile pulita e cibo sano	Italian	No
Initiatives	Clean Drinking Water and Healthy Food	Iniziativa per l'acqua potabile	Italian	No
Initiatives	Clean Drinking Water and Healthy Food	Iniziativa sull'acqua potabile	Italian	No
Initiatives	Save Switzerland from synthetic pesticides	Per una Svizzera senza pesticidi sintetici	Italian	No
Initiatives	Save Switzerland from synthetic pesticides	Iniziativa contro i pesticidi	Italian	No
Initiatives	Save Switzerland from synthetic pesticides	Iniziativa contro pesticidi	Italian	No

Initiatives	Save Switzerland from synthetic pesticides	Iniziativa sui pesticidi	Italian	No
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* Note that the French word pesticide is identical to the English word for pesticide.

Remark: The reason for higher search volumes in German than in French and Italian in Switzerland can be explained by the share these being peoples main language of 63%, 22.7%, 8.1, respectively (FDFA 2020).

Table A2: Important political activities and report releases related to pesticides and initiatives in Switzerland.

Type	Name	Date	Source
Political activity	Start of collecting signatures (Clean Drinking Water and Healthy Food)	2017-03-21	FCh (2020)
Political activity	Initiative submission (Clean Drinking Water and Healthy Food)	2018-01-18	FCh (2020)
Political activity	Publication of the official response by the Swiss Federal Council (Clean Drinking Water and Healthy Food)	2018-12-14	FCh (2020)
Political activity	Start of collecting signatures (Save Switzerland from synthetic pesticides)	2016-11-29	FCh (2020)
Political activity	Initiative submission (Save Switzerland from synthetic pesticides)	2018-05-25	FCh (2020)
Political activity	Publication of the official response by the Swiss Federal Council (Save Switzerland from synthetic pesticides)	2019-02-27	FCh (2020)
National report	Swiss Federal Institute of Aquatic Science and Technology (EAWAG)	2017-04-01	Langer et al. 2017
National report	Swiss Federal Institute of Aquatic Science and Technology (EAWAG)	2019-04-01	Spycher et al. 2019
National report	Swiss Association of Cantonal Chemists (ACCS)	2019-09-06	ACCS 2019
International report	World Health Organization (WHO)	2015-03-20	WHO (2016)
International report	World Health Organization (WHO) & Food and Agriculture Organisation (FAO)	2016-05-16	WHO and FAO (2017)

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8.10 Appendix 2

Google Trends does provide relative amounts of searches compared to the highest search volume at one point in time in the time series. Google allows downloading up to five search terms at once. By the following steps we obtained a common dataset of all search terms (>5): First, we downloaded separately all data for pesticides and initiatives (main groups) related search terms and combined the datasets ($\text{search volume}_{\text{search term, main group, time}}$). Second, we obtained the search term with the highest search volume per main group. Third, we downloaded Google Trends data based on these two search terms, which provide the relative weights that are used for rescaling ($\text{rescale value1}_{\text{main group}}$). Fourth, we rescaled the search volume by the main group rescale values obtained in step 3 ($\text{search volume rescaled}_{\text{main group, time}} = \text{search volume}_{\text{search term, main group, time}} / 100 \times \text{rescale value1}_{\text{main group}}$).

After obtaining a common dataset of all search terms, we computed the main group search volume (ranging from 0 to 100) and the subgroup search volume (also ranging from 0 to 100). Main group search volume: First, we computed the sum of the search volumes of all search terms per main group and date ($\text{sum group}_{\text{main group, time}}$). Second, we identified the maximum sum over all groups and time, which provides the rescale values (rescale value2). Third, we rescaled the value obtained in step one ($\text{group search volume}_{\text{main group, time}} = \text{sum group}_{\text{main group, time}} / \text{rescale value2} \times 100$). Subgroup search volume: The rescaled search volume was likewise obtain as for the main group, except for grouping by subgroup instead of main group ($\text{group search volume}_{\text{subgroup, time}} = \text{sum group}_{\text{subgroup, time}} / \text{rescale value3} \times 100$).

8.11 Appendix 3

In order to address sampling noise and to obtain the historical data we followed the procedure described Carrière-Swallow and Labbé (2013). We downloaded on 33 days (between 18th of January and 24 of February 2020) Google Trends data, from which we calculated the mean search volume over time and the deviation from the trend (Figure A1). Overall search volume did not change decisively between downloads, indicating the adequacy of the data. Note also that deviation from the mean were smaller than for other more popular searches (e.g. Carrière-Swallow and Labbé (2013)).

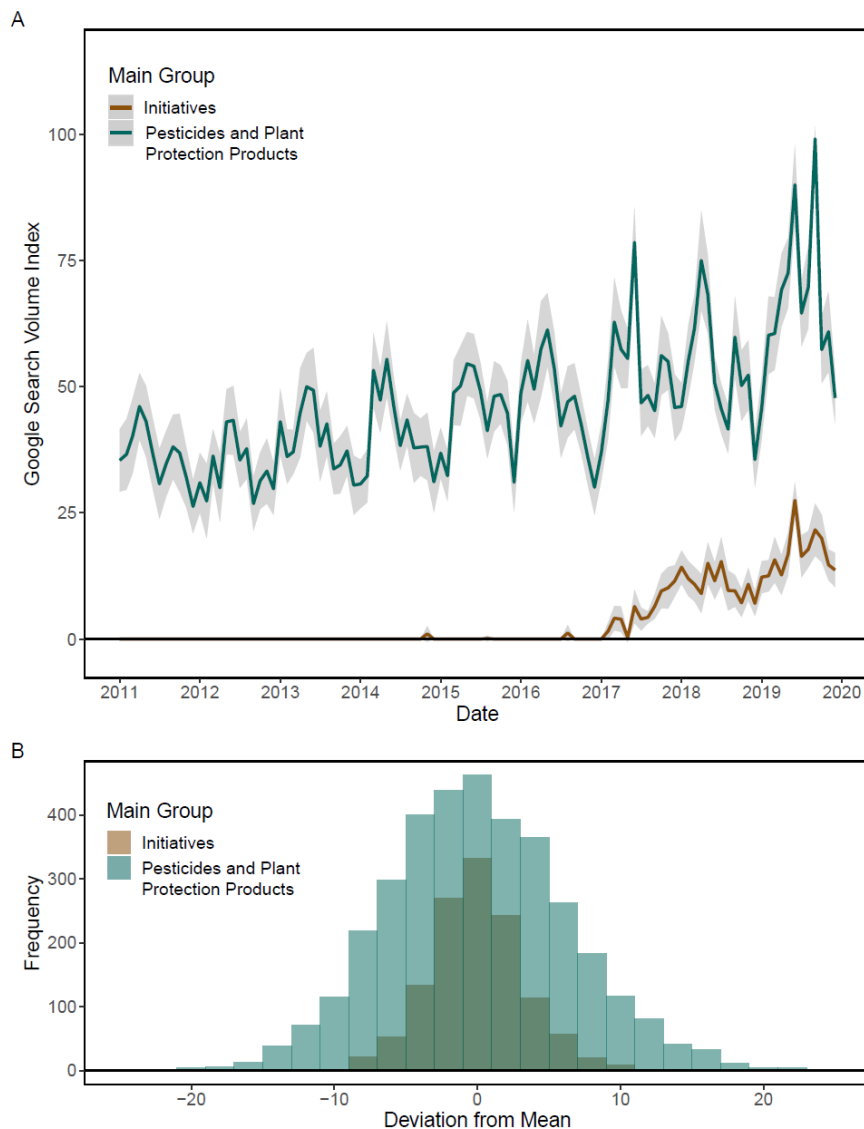


Figure A1: Sampling noise of Google Trends data. Panel A indicates the mean search volume per group and the standard deviation over time. Panel B indicates the deviation from the mean per group.

Carrière-Swallow, Y. & Labbé, F. (2013). Nowcasting with Google Trends in an emerging market. *Journal of Forecasting*, **32**, 289-298. <https://doi.org/10.1002/for.1252>

8.12 Appendix 4

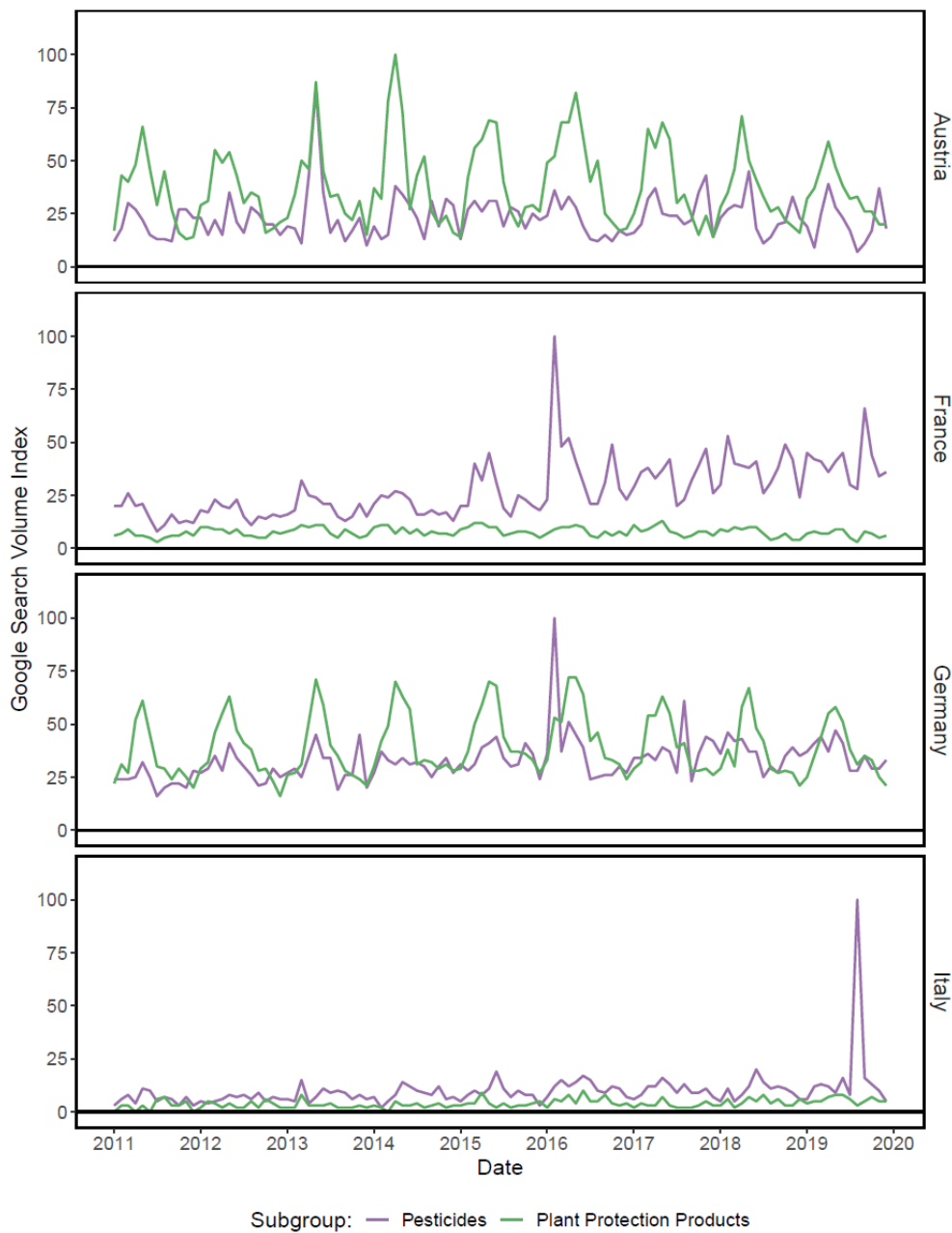


Figure A2: Interest over time for search terms related to pesticides and plant protection products in Austria, France, Germany and Italy. We used the following search terms: 'Pflanzenschutzmittel' and 'Pestizide' for Austria and Germany; 'Produit phytosanitaire' and 'Pesticide' for France and 'Prodotto fitosanitario' and 'Pesticida' for Italy.