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Inhalt

Im Gegensatz zu den in englischer Sprache geschriebenen Publikationen ist der Rahmentext dieser Dissertation bewusst auf Deutsch verfasst, um die publikationsübergreifende Synthese den vor allem in der deutschsprachigen Schweiz aktiven Akteuren zugänglich zu machen. Zu diesem Zweck wurde auch bereits eine Zusammenfassung eines Teils dieser Arbeit in französischer und deutscher Sprache verfasst, die in einer Fachzeitschrift in der Schweiz veröffentlicht wurde. Dieser Artikel wird ebenfalls im Publikationsteil aufgeführt, auch wenn in der Rahmenschrift aufgrund des zusammenfassenden Charakters auf diese Publikation nicht weiter eingegangen wird. Um eine bessere Lesbarkeit zu gewährleisten, wurde bei den meisten Formulierungen auf eine geschlechterdifferenzierende Schreibweise verzichtet. Sämtliche Bezeichnungen wie „Landwirte“, „Teilnehmer“ usw. sind geschlechtsneutral aufzufassen und sprechen alle weiblichen, männlichen und diversen Personen gleichberechtigt an.

Abkürzungen

BIO	Biologisch
BIOBIO	Biodiversity indicators for organic and low-input farming systems
bzw.	beziehungsweise
ca.	zirka
CBD	Convention on Biological Diversity
etc.	et cetera
EU	Europäische Union
IP	Integrierte Produktion
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
ISO	International Standard Organisation
kg	Kilogramm
KMO	Kaiser-Meyer-Olkin-Kriterium
LCA	Life Cycle Assessment
MA	Millennium Ecosystem Assessment
MVP	Mit Vielfalt punkten
ÖLN	Ökologischer Leistungsnachweis
SALCA	Swiss Agricultural Life Cycle Assessment
SMART	Sustainability Monitoring and Assessment RouTine
UN	United Nations
usw.	und so weiter
z. B.	zum Beispiel

Kurzfassung

Diese kumulative Dissertation analysiert verschiedene Strategien zur Steigerung der Biodiversität in der Schweizer Landwirtschaft. Der hier vorgelegte Text bildet den übergreifenden Rahmen für vier begutachtete Forschungsarbeiten, die zwischen 2016 und 2018 veröffentlicht wurden. Leitgebend für die Forschung waren dabei die folgenden Fragen: Wie kann Biodiversität in der Landwirtschaft bewertet werden, und wie vertrauenswürdig sind die bestehenden Methoden, die üblicherweise angewandt werden? Welche Einstellungen und Motivationen führen Landwirte dazu, umweltfreundlichere Produktionssysteme zu wählen und/oder Maßnahmen zur Erhaltung der Biodiversität auf landwirtschaftlichen Betrieben durchzuführen? Welchen Einfluss können Biodiversitätsberatungsdienste auf die Einstellungen und Motivationen der Landwirte haben?

Die Erhaltung und/oder Verbesserung der Biodiversität auf landwirtschaftlichen Betrieben erfordert zuverlässige Methoden zur Biodiversitätsbewertung. Aus diesem Grund liegt ein Schwerpunkt dieser Dissertation auf dem Vergleich verschiedener Methoden, die zur Beurteilung der Auswirkungen der landwirtschaftlichen Produktion auf Biodiversität eingesetzt werden können. Als Referenz für den Vergleich verschiedener Methoden in einer Fallstudie diente eine Untersuchung von Pflanzen, Heuschrecken, Vögeln und Schmetterlingen, die auf fünf landwirtschaftlichen Betrieben über einen Zeitraum von sechs Jahren durchgeführt wurde. Drei verschiedene Biodiversitätsbewertungsmethoden, darunter auch eine im Rahmen der Umweltbewertungsmethode „Ökobilanz", wurden dann auf denselben fünf Betrieben angewandt, um die Möglichkeiten der verschiedenen Methoden und Ansätze zu vergleichen und ihre Vor- und Nachteile zu ermitteln.

Eine Literaturübersicht bestehender Biodiversitätsbewertungsmethoden innerhalb der Ökobilanzen zeigte eine Diskrepanz zwischen Methoden, die bereits global anwendbar, aber zu grob sind, um zwischen verschiedenen landwirtschaftlichen Intensitäten zu unterscheiden, und Methoden, bei denen eine solche Differenzierung zwar möglich ist, die aber auf ein lokales geographisches Gebiet beschränkt bleiben. Die Untersuchungen in der Fallstudie zeigten, dass mit keiner der Methoden eine umfassende Aussage über die Biodiversität eines bestimmten Betriebes gemacht werden konnte, die Methoden aber einzelne wertvolle Informationen über einen oder mehrere Aspekte der Biodiversität liefern konnten. Bei der Erstellung einer Rangliste des Biodiversitätsstatus der Betriebe führten die verschiedenen Methoden zu unterschiedlichen Ergebnissen, ein Umstand, der die Notwendigkeit unterstreicht, bei der Interpretation die Grenzen der jeweiligen Biodiversitätsbewertungsmethoden zu berücksichtigen.

Bei der Auseinandersetzung mit den Forschungsfragen über den Zusammenhang zwischen den Einstellungen und Motivationen der Landwirte und der Biodiversitätsförderung ergaben sich signifikante Korrelationen. Es zeigte sich, dass Landwirte mit und ohne Umweltlabel sich in ihren Naturschutzmotivationen unterscheiden, nicht jedoch in ihren wirtschaftlichen Beweggründen. Ein stärkeres Vertrauen in ein Biodiversitätskonzept, mit dem ein Gewinn für die landwirtschaftliche Produktion möglich ist, war mit der Tendenz verbunden, auf den Betrieben Biodiversitätsförderflächen einzurichten, und stellte gleichzeitig eine Motivation für die Wahl eines Produktionssystems mit Umweltlabel dar. Darüber hinaus zeigten die Ergebnisse, dass eine gezielte Biodiversitätsberatung durch Beratungsdienste die Überzeugung vom Nutzen der Biodiversität auf den Betrieben und die Motivation, entsprechende Maßnahmen zu ergreifen, positiv beeinflussen konnten. Es zeigte sich, dass finanzielle Motivationen unabhängig von intrinsischen Motivationen sind, weshalb in Zusammenarbeit mit den Beratungsdiensten für Landwirte soziale und psychologische Motivationsstrategien eingesetzt werden sollten, um Direktzahlungssysteme zu ergänzen und dadurch die Förderung der Biodiversität auf den landwirtschaftlichen Betrieben zu verbessern.

Abstract

This cumulative dissertation analyses various strategies for increasing biodiversity in Swiss agriculture. The text presented here represents the overarching framework for the four peer-reviewed research papers that were published between 2016 and 2018. The guiding principles of the research are expressed in the following questions: How can on-farm biodiversity be evaluated and how trustworthy are the existing methods that are commonly used? Which beliefs and motivations held by farmers lead them to voluntarily enter environmental labelling schemes and/or implement on-farm actions to conserve biodiversity? What influence can biodiversity advisory services have on farmers' beliefs and motivations?

Addressing questions about on-farm biodiversity preservation and/or enhancement requires a reliable method of assessing biodiversity outcomes. Therefore, a focus of this dissertation is on the comparison of different biodiversity assessment methods that can be used to evaluate the impact of agricultural production on biodiversity. The reference point for the comparison of biodiversity assessment methods in a case study was a survey of plants, grasshoppers, birds, and butterflies that was conducted on five different farms over a six-year period. Three different biodiversity assessment methods, including the biodiversity assessment in the environmental assessment method "life cycle assessment (LCA)", were then applied to these same five farms in order to compare the possibilities of different methods and approaches and to identify their advantages and disadvantages.

A review of commonly used biodiversity assessment methods within LCAs revealed a discrepancy between existing methods that are already globally applicable but too coarse to distinguish between different levels of agricultural intensity and methods for which such differentiation is possible but which are limited to a local geographical area. The case study investigation showed that none of the methods could provide a comprehensive statement on the biodiversity of a particular farm, but each could provide valuable information on one or more aspects of biodiversity. When establishing a ranking of the biodiversity status of the farms, different methods led to different conclusions, which highlights the imperative to consider the limitations of the respective biodiversity assessment methods when interpreting the results.

Addressing the research questions about the relationships between biodiversity outcomes and the beliefs and motivations of farmers revealed significant correlations. Farmers with and without environmental labels were found to differ in their nature conservation motivations, but not in their economic motivations. Stronger beliefs in the concept of biodiversity as an asset that enables agricultural production was associated with a tendency to establish biodiversity promotion areas on the farms along with motivating increased membership of labelling schemes. Furthermore, the results showed that targeted biodiversity advice from advisory services could indeed positively influence beliefs in the benefits of on-farm biodiversity and motivations to take actions.

Financial motivation was found to be independent of intrinsic motivation. Thus, social and psychological motivation strategies, in collaboration with farmer advisory services, should be used to supplement direct payment schemes and thereby enhance the promotion of biodiversity on farm

1 Einleitung

Der global fortschreitende Biodiversitätsverlust stellt eine massive Umweltbedrohung dar, deren Bedeutung in den letzten Jahren zunehmend in den Fokus gerückt ist (Rockstrom *et al.*, 2009; Steffen *et al.*, 2015; Newbold *et al.*, 2016). Als Reaktion auf den Artenschwund und die Biotopentwertung wurde bereits im Jahr 1992 ein Biodiversitätsübereinkommen (*Convention on Biological Diversity* [CBD]) getroffen, das inzwischen von 168 einzelnen Staaten und der EU getragen wird. Mit nationalen Strategien und Aktionsplänen wird versucht, dem Biodiversitätsschwund entgegenzuwirken (*Secretariat of the Convention on Biological Diversity* 2014; Sarkki *et al.*, 2016). Dennoch ist es bislang nicht gelungen, den Rückgang der Biodiversität aufzuhalten oder zu verlangsamen (Butchart *et al.*, 2010; Carvalheiro *et al.*, 2013). Auch für das Jahr 2018 wird der stetige Verlust an Biodiversität vom Weltbiodiversitätsrat (IPBES) bestätigt (Pauli, 2018). Es ist seit langem bekannt, dass die Landwirtschaft ein Treiber dieser Entwicklung ist (MA, 2005; Butler *et al.*, 2007) und ihr deswegen im Bemühen gegen den Artenverlust besondere Bedeutung zukommt (Pe'er *et al.*, 2014). Weltweit werden etwa fünf Milliarden Hektar Land landwirtschaftlich genutzt, was etwa 37 % der gesamten globalen Landfläche entspricht (Umweltbundesamt, 2013). Die ausgedehnte Flächennutzung zu Lasten ehemals natürlicher bzw. naturnaher Biotope und die Intensität der landwirtschaftlichen Produktion haben dazu geführt, dass viele Landschaftselemente verschwunden und damit deren Funktionen als wertvolle Habitate verlorengegangen sind (Billeter *et al.*, 2008). In intensiv genutzten Regionen werden solche Habitate heute nur selten angemessen erhalten (Hole *et al.*, 2005).

Mit verschiedenen Instrumenten wird versucht, den negativen Einfluss der Landwirtschaft auf Biodiversität zu verringern. In vielen Ländern wird dies in Form von Agrarumweltmaßnahmen umgesetzt.

Dabei erhalten Landwirte Gelder als Gegenleistung für eine Biodiversitätsförderung auf ihren Betrieben. So müssen in der Schweiz beispielsweise 7 % der landwirtschaftlichen Nutzfläche im landwirtschaftlichen Betrieb als Biodiversitätsförderfläche ausgewiesen werden, um Direktzahlungen zu erhalten (Lanz und Lehmann, 2012). Des Weiteren werden ähnlich wie in der EU auch in der

Schweiz von der Regierung finanzielle Anreize gesetzt, um halbnatürliche Habitate auf den Betrieben zu erhalten (BLW, 2013).

Mit dieser Arbeit werden zwei Themenkomplexe im Spannungsfeld zwischen Landwirtschaft und Biodiversität näher beleuchtet, um Strategien zur Steigerung der Biodiversitätsförderung in der Landwirtschaft zu entwickeln. Zum einen soll dabei die zentrale Funktion der Landwirte untersucht werden, um herauszufinden, welche Motivationen und Einstellungen entscheidend sind, um auf den jeweiligen Betrieben Biodiversitätsförderung zu betreiben. Dabei werden die Möglichkeit der Biodiversitätsberatung und der Einfluss verschiedener Produktionssysteme in die Betrachtung einbezogen. Zum anderen werden Methoden verglichen, die dazu dienen können, die Auswirkung der Landwirtschaft auf Biodiversität zu bewerten. Dazu werden zunächst in einer Literaturübersicht die aktuellen Entwicklungen bezüglich der Umweltbewertungsmethode der Ökobilanz zusammengefasst. Danach werden im Rahmen einer Fallstudie verschiedene Biodiversitätsbewertungsmethoden, darunter auch eine Ökobilanzmethode, auf fünf landwirtschaftlichen Betrieben angewandt und einander gegenübergestellt.

2 Forschungsfragen

Im Fokus dieser Arbeit steht die Biodiversitätsförderung im landwirtschaftlichen Betrieb. Die zwei Themenkomplexe (a, b) wurden mit Hilfe von sechs Forschungsfragen näher untersucht.

a) Welche Motivationen und Einstellungen leiten die Landwirte bei der Wahl der Biodiversitätsförderung, und welchen Einfluss haben das Instrument der Biodiversitätsberatung und das Produktionssystem auf die Bereitschaft der Landwirte, Biodiversitätsförderung einzusetzen / zu betreiben?
b) Welche Biodiversitätsbewertungsmethoden sind geeignet, um die Auswirkungen der landwirtschaftlichen Produktion auf Biodiversität erfassen zu können?

Folgende Forschungsfragen wurden bearbeitet:

1) Inwieweit können Einstellungs- oder Motivationsfaktoren die biodiversitätsrelevanten Umweltentscheidungen (Bereitstellung ökologischer Ausgleichsflächen) der Schweizer Landwirte beeinflussen? (Publikation I)

2) Unterscheiden sich die Motivationen und Einstellungen von Landwirten unterschiedlicher Produktionssysteme? (Publikationen I und II)

3) Ist es möglich, durch Biodiversitätsberatung die Motivationen und Einstellungen der Landwirte bezüglich Biodiversitätsfördermaßnahmen zu verändern? (Publikation II)

4) Inwieweit sind Biodiversitätsbewertungsmethoden in Ökobilanzen dazu geeignet, die Biodiversitätswirkung der landwirtschaftlichen Produktion zu bewerten? (Publikation III)

5) Wie unterscheiden sich die verschiedenen Biodiversitätsbewertungsmethoden in ihrer Anwendung und welche Vor- und Nachteile zeigen sich bei den verschiedenen Methoden in ihrer Anwendbarkeit? (Publikation IV)

6) Eignen sich die jeweiligen Methoden vor dem Hintergrund ihres zugedachten Verwendungszwecks für die Biodiversitätsbewertung landwirtschaftlicher Betriebe? (Publikation IV)

3 Publikationen

In diesem Kapitel werden die Publikationen I-IV in einer kurzen Zusammenfassung vorgestellt. Zu diesem Zweck werden auch die jeweiligen Forschungsfragen nochmals aufgeführt. Zudem wird der Bezug zwischen den verschiedenen Publikationen hergestellt.

3.1 Publikation I: Gabel, V.M., Home, R., Stolze, M., Pfiffner, L., Birrer, S., Köpke, U., 2018. Motivations for swiss lowland farmers to conserve biodiversity: Identifying factors to predict proportions of implemented ecological compensation areas. Journal of Rural Studies 62, 68-76.

Diese Publikation widmet sich der grundlegenden Frage, welche Einstellungen und Motivationen der Landwirte Auswirkung auf die Biodiversitätsförderung auf den jeweiligen Betrieben haben.

Forschungsfrage 1: Inwieweit können Einstellungs- oder Motivationsfaktoren die biodiversitätsrelevanten Umweltentscheidungen (Bereitstellung ökologischer Ausgleichsflächen) der Schweizer Landwirte beeinflussen?

Forschungsfrage 2: Unterscheiden sich die Motivationen und Einstellungen von Landwirten unterschiedlicher Produktionssysteme?

In der wissenschaftlichen Forschung geht man häufig von einem methodischen Rahmen aus. In unserer Analyse in Publikation I hat sich die „Theorie des überlegten Handelns“ (Ajzen, 1991) als vielversprechender Theorieansatz gezeigt, um die Daten einzuordnen und zu interpretieren (für weitere Erläuterungen zum methodischen Hintergrund siehe Anhang B).

Ziel dieser quantitativen Studie war es, herauszufinden, welchen Einfluss Einstellungs- und Motivationsfaktoren auf umweltbewusste Entscheidungen von Schweizer Landwirten haben. Konkret behandelt die Publikation die Anlage von Biodiversitätsförderflächen auf den jeweiligen landwirtschaftlichen Betrieben. Dabei wurde untersucht, welche Motivationen und Einstellungen der Landwirte die Umsetzung dieser Biodiversitätsfördermaßnahme beeinflussen. Ein zusätzlicher Aspekt, der dabei näher untersucht wurde, war die Frage, ob es Einstellungs- und

Motivationsunterschiede zwischen Biolandwirten und konventionellen Landwirten gibt. Basierend auf einer Literaturrecherche und einigen qualitativen Interviews (Home *et al.* 2014) wurden zwei Skalen entwickelt, in denen jeweils Einstellungs- und Motivationsaussagen abgefragt wurden. Im Weiteren werden diese „Einstellungsskala“ und „Motivationsskala“ genannt. Insgesamt wurden 133 Landwirte mit Hilfe eines Fragebogens befragt. 60 davon waren Landwirte, die nach Biorichtlinien produzieren. Die Ergebnisse haben gezeigt, dass es signifikante Korrelationen zwischen verschiedenen Einstellungs- und Motivationsfaktoren und der tatsächlich etablierten Biodiversitätsförderfläche auf den Betrieben gibt. Bei den Einstellungen korrelierten die folgenden Aussagen mit der Biodiversitätsfläche auf den Betrieben: *„Die Erhaltung und Förderung der Biodiversität ist eine wichtige Aufgabe“, „Die Förderung der Biodiversität ist für meinen Betrieb wichtig“, „Die Bereitstellung und Erhaltung von Erholungsgebieten ist eine wichtige Aufgabe“, „Ich denke, die breite Öffentlichkeit schätzt unsere Arbeit zur Förderung der biologischen Vielfalt“ und „Nach meiner Erfahrung haben sich Ökosystemleistungen wie Bestäubung und Schädlingsbekämpfung durch das Anlegen von Biodiversitätsförderflächen verbessert“.* Je größer die Zustimmung der Landwirte zu den hier aufgeführten Aussagen war, desto größer war der Anteil der von ihnen angelegten Biodiversitätsförderfläche auf dem eigenen Betrieb. Bei sieben Einstellungsaussagen gab es zudem eine höhere Zustimmung der Biolandwirte. Bei der Abfrage nach den Motivationen zur Anlage von Biodiversitätsförderflächen gab es eine signifikante Korrelation zwischen den folgenden Aussagen und der Biodiversitätsförderfläche: *Ich lege Biodiversitätsförderflächen an, … „weil ich zur Erhaltung und Förderung der Biodiversität beitragen möchte“, „weil ich dafür Direktzahlungen erhalte“, „weil ich zu einer diversen Agrarlandschaft beitragen möchte“, „weil ich zu einem guten Image der Landwirtschaft beitragen möchte“, „weil ich innerlich überzeugt bin“, „weil es zur Produktionsphilosophie auf dem Betrieb passt“, „weil sie Sinn für die Betriebsstruktur machen“, „weil ich Land dafür nutzen kann, das weniger gut für die Produktion geeignet ist“, „weil sie praktisch sind und die Produktion nicht beeinträchtigen“, „weil ich dadurch eine ökologische Leistung erhalte“* und *„weil sie einen positiven Effekt auf meinen Produktabsatz haben“.* Bei den Motivationsaussagen war die Zustimmung der Biolandwirte in fünf Fällen ausgeprägter; alle diese Aussagen waren umweltorientiert.

Durch die Überprüfung von Korrelationen zwischen den verschiedenen Faktoren hat sich gezeigt, dass die finanziellen Motivationen unabhängig von den intrinsischen Motivationen sind und damit in der bekannten Theorie des überlegten Handelns von Ajzen (1991) eher einer „wahrgenommenen“ als einer „verhaltensorientierten“ Kontrolle zugeordnet werden können. Das galt sowohl für Biolandwirte als auch für ihre konventionellen Kollegen. Aus diesem Ergebnis konnte abgeleitet werden, dass eine Kombination aus finanziellen Anreizen und einer Steigerung der intrinsischen Motivationen erfolgversprechend ist, um ein optimales Ergebnis bezüglich des Biodiversitätsschutzes auf den Landwirtschaftsbetrieben zu erreichen.

3.2 Publikation II: Gabel, V.M., Home, R., Stolze, M., Birrer, S., Steinemann, B., Köpke, U., 2018a. The influence of on-farm advice on beliefs and motivations for Swiss lowland farmers to implement ecological compensation areas on their farms. The Journal of Agricultural Education and Extension, 1-16.

Nach der Identifizierung der Einstellungen und Motivationen, die einen Einfluss auf die Anlage von Biodiversitätsförderflächen haben, stellte sich die Frage, ob und inwieweit sich diese möglicherweise durch eine gezielte Biodiversitätsberatung beeinflussen lassen. Mit dieser Fragestellung kann überprüft werden, ob sich die Empfehlung, die aus Publikation I abgeleitet wurde, nämlich die intrinsische Motivation der Landwirte zu steigern, durch eine Beratungsleistung erzielen lässt. Zu diesem Zweck wurde mit demselben Forschungsdesign wie in Publikation I der Unterschied zwischen Landwirten mit und ohne Biodiversitätsberatung untersucht. Auch hier wurde die Zugehörigkeit zu unterschiedlichen Produktionssystemen (mit einem Umweltlabel [BIO, IP] und ohne Umweltlabel) in die Betrachtung miteinbezogen.

Forschungsfrage 2: Unterscheiden sich die Motivationen und Einstellungen von Landwirten unterschiedlicher Produktionssysteme?

Forschungsfrage 3: Ist es möglich, durch Biodiversitätsberatung die Motivationen und Einstellungen der Landwirte bezüglich Biodiversitätsfördermaßnahmen zu verändern?

Die Datenerhebung und die Stichprobe in dieser Publikation basieren ebenfalls auf der Fragebogenerhebung und sind damit identisch mit Publikation I. Auch hier wurden die „Einstellungs“- und „Motivationsskala“ genutzt, und als statistische Methode wurde zur Ermittlung der Gruppenunterschiede auf den Mann-Whitney-U-Test zurückgegriffen (für weitere Erläuterungen zum methodischen Hintergrund siehe Anhang B).

Das Ziel dieser Studie war es, das Potential der Biodiversitätsberatung auf landwirtschaftlichen Betrieben abzuschätzen, um zu erkennen, ob mit diesem Instrument die Einstellungen und Motivationen der Landwirte zur Biodiversitätsförderung auf ihren Betrieben beeinflusst werden können. Unter den 133 Befragten befanden sich 23 Landwirte, die über einen Zeitraum von 6 Jahren regelmäßig Biodiversitätsberatungen durch Experten auf ihren Betrieben erhalten hatten. Zur Beantwortung einer weiteren Fragestellung wurde untersucht, ob sich Landwirte verschiedener Produktionssysteme (BIO, IP und konventionell) in ihren Einstellungen und Motivationen bezüglich der Biodiversitätsförderung unterscheiden. Zur Beantwortung der Fragen wurde wieder eine fünfstufige Likert-Skala verwendet. Viele Fragen in Bezug auf die Einstellungsskala, die sich auf die Aufgaben der Landwirte konzentrieren, haben von allen Teilnehmenden eine hohe Zustimmung bekommen, unabhängig davon, welches Produktionssystem sie vertreten und ob sie eine Biodiversitätsberatung erhalten hatten oder nicht. Darunter waren z. B. die „*Erhaltung der Bodenfruchtbarkeit*“, die „*Produktion von gesunden Nahrungsmitteln*“ oder ein „*tierfreundlicher Umgang*“. Signifikante Unterschiede zwischen beratenen und nicht beratenen Landwirten waren eher unter den Einstellungen zu finden, die insgesamt weniger Zustimmung erhalten haben. So haben die beratenen Landwirte eher zugestimmt, *„dass es für sie keinen Widerspruch zwischen Produktion und Biodiversitätsförderung gibt“, „dass eine Biodiversitätsförderung für den Betrieb wichtig ist“* und *„dass die Förderung der Biodiversität von der Gesellschaft wertgeschätzt wird“.* Die Landwirte mit Umweltlabel zeigten eine signifikant größere Zustimmung bei allen Einstellungsfaktoren außer bei den Aussagen, *„dass die Sicherstellung einer sicheren Versorgung der Schweizer Bevölkerung mit Lebensmitteln eine wichtige Aufgabe der Schweizer Landwirtschaft darstellt“* und *„dass die Bereitstellung und Pflege von Erholungsräumen eine wichtige Aufgabe ist“*. Bei den Motivationsaussagen wurden insgesamt mehr Unterschiede zwischen den beratenen und nicht beratenen Landwirten festgestellt. Die beratenen Teilnehmer stimmten demnach eher zu, Biodiversitätsförderflächen anzulegen, … *„weil sie glauben, dass die Landwirtschaft eine Umweltverantwortung trägt“, „weil sie einen Beitrag zur Förderung und Erhaltung der Biodiversität leisten wollen“, „weil sie zu einer diversen Agrarlandschaft beitragen wollen“, „weil sie innerlich überzeugt sind“, „weil es zur Produktionsphilosophie auf ihrem Betrieb passt“* und *„weil sie praktisch sind und die Produktion nicht beeinträchtigen“.* Auch bei den Motivationen war der Unterschied zwischen der Gruppe mit und jener ohne Umweltlabel noch deutlicher, und die Landwirte mit Umweltlabel gaben bei zehn Motivationsfaktoren eine größere Zustimmung an. Keine Unterschiede konnten bei den folgenden Motivationen festgestellt werden*: Ich lege Biodiversitätsförderflächen an, … „weil ich dafür Direktzahlungen erhalte“, „weil ich zu einem guten Bild der Landwirtschaft beitragen möchte“, „weil sie aus der Betriebsstruktur heraus Sinn machen“, „weil sie ästhetisch aussehen“* und *„weil so Land genutzt werden kann, das sich nicht zur Produktion eignet“.* Die Ergebnisse der Publikation haben gezeigt, dass auch sogenannte Schlüsseleinstellungen und Motivationen, die eine Korrelation mit der tatsächlichen Biodiversitätsförderfläche aufwiesen, durch die Beratungstätigkeit beeinflusst werden konnten. Beratene Landwirte stimmten demnach eher zu, *„dass Biodiversität wichtig ist“,* „dass *Biodiversitätsschutz und landwirtschaftliche Produktion kompatibel sind“* und „*dass Naturschutz auf landwirtschaftlichen Betrieben von der Gesellschaft erwünscht ist*“.

3.3 Publikation III: Gabel, V.M., Meier, M.S., Köpke, U., Stolze, M., 2016. The challenges of including impacts on biodiversity in agricultural life cycle assessments. Journal of Environmental Management 181, 249-260.

Eine wichtige Erkenntnis, die sich aus den Publikationen I und II ableiten lässt, war das Bedürfnis der Landwirte, wirkungsvoll zu agieren. Sie wollen nur solche Biodiversitätsfördermaßnahmen umsetzen, die tatsächlich zu einer Verbesserung der Biodiversität auf ihren Betrieben führen. Um zu überprüfen, ob eine Biodiversitätsmaßnahme den erwünschten Erfolg gebracht hat oder um einen Vergleichszustand (vorher-nachher oder zwischen verschiedenen Betrieben) herstellen zu können, werden Biodiversitätsbewertungsmethoden benötigt.

Eine mögliche Methode ist die Bewertung der Biodiversität als Kategorie der Umweltbewertungsmethode „Ökobilanz“. Dementsprechend sollte geprüft werden, ob die vorhandenen Biodiversitätsbewertungsmethoden und -ansätze für Ökobilanzen bereits eingesetzt werden können.

> Forschungsfrage 4: Inwieweit sind Biodiversitätsbewertungsmethoden in Ökobilanzen dazu geeignet, die Biodiversitätswirkung der landwirtschaftlichen Produktion zu bewerten?

Um einen Überblick über die bestehenden Methoden zu geben, wurde das Verfahren der systematischen Literaturübersicht gewählt (für weitere Erläuterungen zum methodischen Hintergrund siehe Anhang B).

Die bestehenden Methoden und Methodenansätze für eine Biodiversitätsbewertung innerhalb von Ökobilanzen wurden zusammengetragen und die jeweiligen Methoden hinsichtlich ihrer Anwendbarkeit in landwirtschaftlichen Ökobilanzen analysiert. Insgesamt wurden die Stärken und Schwächen von 22 Methoden und Methodenansätzen in die Analyse einbezogen. Die Analyse beinhaltete verschiedene Kriterien, die sich aus den Anforderungen der Biodiversitätsbewertung im Allgemeinen, bestimmten Ökobilanzmethodenanforderungen sowie aus Besonderheiten in der Betrachtung landwirtschaftlicher Produkte ergaben. Im Fokus standen dabei die Wahl der funktionellen Einheit, die gewählten Biodiversitätsaspekte, Biodiversitätsindikatoren und Referenzsituationen. Hinzu kamen die Möglichkeit, die Methoden grundsätzlich weltweit anzuwenden, und die Fähigkeit, verschiedene landwirtschaftliche Produktionsintensitäten zu unterscheiden. Die Methodenentwicklung, die in dieser Publikation betrachtet wird, erstreckt sich über einen Zeitraum von ca. 15 Jahren, von 2000 bis 2015. Dabei gab es sowohl Methoden, auf denen durch Weiterentwicklung aufgebaut wurde, als auch Methoden, die sich unabhängig von vorherigen Entwicklungen eigenständig präsentieren. Nur sehr wenige der Methoden wurden im Hinblick auf eine Eignung zur Bilanzierung von landwirtschaftlichen Produkten oder Produktionssystemen entwickelt. Es wurde eine große Diskrepanz festgestellt zwischen Methoden, die bereits jetzt weltweit einsetzbar sind, aber gleichzeitig zu grob sind, um verschiedene landwirtschaftliche Intensitäten zu differenzieren, und Methoden, mit denen eine solche Differenzierung zwar möglich ist, die sich aber auf einen begrenzten geographischen Raum beziehen und dabei eine sehr hohe Datenmenge erfordern.

Bei der Kriterienanalyse zeigte sich, dass bei der Wahl der funktionellen Einheit bei der Hälfte der Studien ein Bezug zur Fläche hergestellt wurde, während die andere Hälfte eine produktbezogene funktionelle Einheit wählte. Der am häufigsten abgedeckte Biodiversitätsaspekt war die Artenvielfalt, gefolgt von der Biotopqualität und seltenen bzw. gefährdeten Arten. In zwei Fällen wurde auch der Aspekt der funktionellen Biodiversität betrachtet. Die genetische Diversität wurde hingegen von keiner der analysierten Methoden berücksichtigt. Für die meisten Methoden wurde keine Begründung für die Auswahl des Biodiversitätsaspektes genannt. Die Gründe für die Auswahl der Biodiversitätsindikatoren waren: *„die Datenverfügbarkeit*", *„der Einsatz der Indikatoren in vorherigen Studien"* und *„die Anerkennung eines Indikators unter Ökobilanzanwendern*". Korrelationen zwischen den abgedeckten Biodiversitätsaspekten und den eingesetzten Biodiversitätsindikatoren waren nicht immer ersichtlich oder klar definiert. Als Referenzsituation wurde in den meisten Fällen ein *„natürlicher oder halbnatürlicher Zustand"* verwendet. Des Weiteren wurden aber auch *„alternative Nutzungsszenarien"*, *„ein Biodiversitätspotential"* oder *„ein erwünschter Zielzustand"* eingesetzt. Eine Methode ist so ausgelegt, dass die Verwendung von verschiedenen Referenzsituationen ermöglicht wird. Eine Grenze dieser Studie wurde darin erkannt, dass der Vergleich der verschiedenen Methoden nur theoretisch erfolgte und damit keine Aussage über die tatsächlichen Anwendungseigenschaften der Methoden getroffen werden konnte.

3.4 Publikation IV: Gabel, V.M., Home, R., Stöckli, S., Meier, M., Stolze M., Köpke U., 2018. Evaluating on-farm biodiversity: A comparison of assessment methods. Sustainability 10 (12), 4812, 1-14.

Aus der Einschränkung heraus, dass es sich bei Publikation III um einen rein theoretischen Vergleich verschiedener Biodiversitätsbewertungsmethoden für Ökobilanzen handelt und aus der daraus abgeleiteten Empfehlung eines praktischen Vergleichs verschiedener Biodiversitätsbewertungsmethoden, wurde der Ansatz für Publikation IV abgeleitet. Dabei sollten verschiedene Biodiversitätsbewertungsmethoden auf mehreren landwirtschaftlichen Betrieben angewandt und verglichen werden.

Forschungsfrage 5: Wie unterscheiden sich die verschiedenen Biodiversitätsbewertungsmethoden in ihrer Anwendung und welche Vor- und Nachteile zeigen sich in ihrer Anwendbarkeit?

Forschungsfrage 6: Eignen sich die jeweiligen Methoden vor dem Hintergrund ihres zugedachten Verwendungszwecks für die Biodiversitätsbewertung landwirtschaftlicher Betriebe?

Um einen Vergleichsansatz durchführen zu können, wurde für Publikation IV die Methode der Fallstudienuntersuchung gewählt (für weitere Erläuterungen zum methodischen Hintergrund siehe Anhang B).

In diesem wissenschaftlichen Artikel werden drei verschiedene Biodiversitätsbewertungsmethoden im Rahmen einer Fallstudie angewandt und analysiert. Alle Methoden wurden auf fünf Betrieben mit unterschiedlichen Produktionssystemen und in zwei verschiedenen Landschaftsräumen angewandt. Für alle Fallstudienbetriebe standen Datenerhebungen von unterschiedlichen Indikatorarten und verschiedene Biodiversitätsmaße zur Verfügung. Es wurden zwei etablierte Methoden, die gezielt für die Biodiversitätsbewertung auf Betriebsebene entwickelt wurden (MVP, SMART), und eine Ökobilanzmethode (LCA-Methode), bei der mit einer Bewertung auf Betriebsebene ein neuer Anwendungsbereich getestet werden sollte, ausgewählt. Ursprünglich wurde die LCA-Methode für eine produktbezogene Bewertung entwickelt. Trotz ihrer unterschiedlichen Vorbedingungen haben alle Methoden den Anspruch, eine Aussage über die Biodiversitätswirkung landwirtschaftlicher Betriebe treffen zu können.

In der Analyse wurden folgende Kriterien näher betrachtet: Biodiversitätsaspekt, Biodiversitätsindikatoren, Referenzsituation, räumliche Bezugsgröße, geographischer Anwendungsbereich, ursprünglicher Verwendungszweck, Anwendungsdauer, Datenverfügbarkeit und die notwendige Expertise der Anwender. Das Ziel dieser Publikation war es, die für verschiedene Zwecke entwickelten Biodiversitätsbewertungsmethoden einander gegenüberzustellen, um ihre jeweiligen Stärken und Schwächen zu identifizieren. Mögliche Unterschiede in den Ergebnissen wurden erläutert und mögliche Einsatzzwecke sowie Anwendungsbereiche der Methoden identifiziert. Für den Vergleich der Methoden wurden die jeweiligen Ergebnisse in einer Reihenfolge vom „besten“ bis hin zum „schlechtesten“ Betrieb geordnet. Keine der Methoden konnte eine ganzheitliche Aussage über den Biodiversitätsstatus der Betriebe liefern. Dennoch konnten die Methoden einen Hinweis oder eine Annäherung an einen oder mehrere Biodiversitätsaspekte liefern. Die Ergebnisse zeigten, dass mit den verschiedenen Biodiversitätsbewertungsmethoden durchaus unterschiedliche Aussagen über den Zustand der Biodiversität auf denselben Betrieben generiert wurden. Damit ist gemeint, dass nicht bei allen Methoden das Biodiversitätsranking der Betriebe die gleiche Reihenfolge ergab. Alle Methoden zeigten Stärken und Schwächen, weshalb bei der Auswahl einer Methode stets das Ziel und damit auch der Einsatzzweck der Biodiversitätsbewertung im Vordergrund stehen sollte. So bietet die SMART-Methode vor allem den Vorteil, dass ein vergleichsweise geringer Arbeitsaufwand benötigt wird, die Methode überall auf der Welt angewandt werden kann und neben den Ergebnissen zur Biodiversität auch noch andere Nachhaltigkeitsaspekte abgedeckt werden. Die LCA-Methode basiert dagegen auf relativ wenigen Einflussparametern und kann ohne einen Betriebsbesuch durchgeführt werden. In ihrem momentanen Entwicklungszustand ist sie aber besonders aufgrund der notwendigen Kartierung der um den Betrieb liegenden Landschaft noch sehr arbeitsintensiv, ein Umstand, der sich aber in Zukunft bei höherer Datenverfügbarkeit deutlich reduzieren könnte. Momentan beschränkt sich die Anwendbarkeit der LCA-Methode noch auf Tieflandgebiete in den gemäßigten Klimaregionen. Hier verdeutlicht sich auch nochmals die grundlegende Problematik zwischen dem Anspruch der Ökobilanz, weltweit einsetzbar zu sein, und der Notwendigkeit, Biodiversität möglichst lokal zu bewerten. Ausschließlich in der Schweiz anwendbar ist das MVP-Punktesystem, das sich stark an den Schweizer Agrarumweltmaßnahmen orientiert. Positiv ist auch hier die kurze Erhebungsdauer. Allerdings wird von den Anwendern Expertise in Biologie und eine genaue Kenntnis des Schweizer Subventionssystems verlangt. Die unterschiedlichen Stärken und Schwächen weisen darauf hin, dass bei der Interpretation der Ergebnisse die jeweiligen Grenzen der gewählten Methoden berücksichtigt werden müssen.

4 Diskussion und Zusammenfassung

Im folgenden Abschnitt werden die Ergebnisse der verschiedenen Untersuchungen vor dem Hintergrund der aufgestellten Forschungsfragen zusammengefasst diskutiert. Im darauffolgenden Kapitel 5 wird mit einem Ausblick ein übergreifendes Fazit gezogen.

Mit dem Aufstellen der ersten Forschungsfrage sollte die Frage geklärt werden, inwieweit Einstellungs- und Motivationsfaktoren biodiversitätsrelevante Umweltentscheidungen (Bereitstellung ökologischer Ausgleichsflächen) von Schweizer Landwirten beeinflussen können.

Die Ergebnisse haben gezeigt, dass es sowohl Einstellungsfaktoren als auch Motivationsfaktoren gibt, die eine Korrelation mit der Biodiversitätsförderfläche aufweisen. Bei der Abfrage verschiedener Einstellungen korrelierten fünf von fünfzehn Faktoren mit der angelegten Biodiversitätsförderfläche. Diese waren: *„Die Erhaltung und Förderung der Biodiversität ist eine wichtige Aufgabe“, „Die Förderung der Biodiversität ist für meinen Betrieb wichtig“, „Die Bereitstellung und Erhaltung von Erholungsgebieten ist eine wichtige Aufgabe“, „Ich denke, die breite Öffentlichkeit schätzt unsere Arbeit zur Förderung der biologischen Vielfalt“* und *„Nach meiner Erfahrung haben sich Ökosystemleistungen wie Bestäubung und Schädlingsbekämpfung durch das Anlegen von Biodiversitätsförderflächen verbessert“.* Auffällig ist, dass es sich dabei um Einstellungen handelt, die eine vergleichsweise geringe allgemeine Zustimmung erhalten haben, obwohl das Zustimmungsniveau insgesamt bei allen Einstellungsvariablen sehr hoch war. Man kann daraus schließen, dass es sich bei den Einstellungen mit hoher Zustimmung aller Befragten nicht um solche handelt, die einen Unterschied in der Entscheidung Biodiversitätsförderflächen anzulegen hervorrufen. Es ist daher davon auszugehen, dass diese Einstellungen eher die allgemeine Wahrnehmung der Umfrageteilnehmer zu ihrem Berufsbild als Landwirte widerspiegeln.

Bei den Motivationsvariablen zeigte sich ein anderes Bild. Dort korrelierten zwölf von fünfzehn Variablen mit der angelegten Biodiversitätsförderfläche und nur bei den folgenden drei Motivationsfaktoren konnte keine Korrelation festgestellt werden:

Ich lege Biodiversitätsförderflächen an, … „weil ich denke, dass die Landwirtschaft eine Umweltverantwortung trägt“, „weil ich mich zukünftigen Generationen gegenüber verantwortlich fühle“, „weil ich davon überzeugt bin, dass die Maßnahmen Sinn machen und effektiv sind“. Insgesamt war das Zustimmungsniveau der Motivationen aber nicht so hoch wie bei den Einstellungen.

Eine weitere wichtige Erkenntnis, die durch diese quantitativen Untersuchungen bekräftigt werden konnte, war, dass bei nahezu allen Landwirten neben einer wirtschaftlichen Motivation eine naturschützende Motivation existiert und dass diese beiden Motivationskomponenten einen Einfluss auf die Entscheidungen und Handlungen haben. Dabei konnte festgestellt werden, dass Direktzahlungen, die häufig einen wichtigen Teil der wirtschaftlichen Stabilität der Betriebe darstellen, sich von den anderen Motivationen, Biodiversität zu fördern, unterscheiden. In einer weiteren Korrelationsanalyse konnte auch kein Zusammenhang zwischen den Schlüsseleinstellungen (Einstellungen, die mit der Biodiversitätsförderfläche korrelieren) und den Direktzahlungen festgestellt werden. Allerdings gab es eine starke Korrelation zwischen der Zustimmung zur Bedeutung der Direktzahlungen und dem Anteil der Biodiversitätsförderfläche auf den Betrieben. In der erweiterten Theorie des überlegten Handelns (Ajzen, 1991) können die Direktzahlungen deswegen eher einer tatsächlichen Verhaltenskontrolle als einer wahrgenommenen Verhaltenskontrolle zugeordnet werden (für weitere Erläuterungen zum methodischen Hintergrund siehe Anhang B). Diese Ergebnisse unterstützen die Forderung nach sozialen und psychologischen Motivationsstrategien als Ergänzung zu den Direktzahlungen.

Die Ergebnisse aus der Untersuchung in der Schweiz können als allgemeines Beispiel für auf Subventionen basierende Agrarumweltprogramme dienen. Sie helfen damit, zu verstehen, wie die Einstellungen und Motivationen der Schweizer Landwirte genutzt werden können, um die Wirksamkeit finanzieller Anreizsysteme auch in anderen Ländern zu verbessern.

Eine weitere Frage, die sowohl in Publikation I wie auch in Publikation II bearbeitet wurde, befasste sich damit, ob es Unterschiede in den Motivationen und Einstellungen von Landwirten gibt, die nach verschiedenen Produktionssystemen wirtschaften.

Dabei konnte in Publikation I bei den umweltbezogenen Motivationen eine signifikant größere Zustimmung von Biolandwirten im Vergleich zu den konventionell wirtschaftenden festgestellt werden. Im Gegensatz dazu wurden keine Motivationsunterschiede zwischen den beiden Produktionssystemen gefunden, die im Zusammenhang mit finanziellen Gewinnen stehen. Ebenso konnte in Betrieben des biologischen Landbaus keine umfänglichere Biodiversitätsförderfläche festgestellt werden.

Zusammenfassend betrachtet, spiegeln sich die umweltbezogenen Einstellungen der Landwirte des ökologischen Landbaus primär in der Wahl des umweltfreundlicheren Produktionssystems wider; gleichzeitig sind Biolandwirte nicht weniger produktionsorientiert, und ihre ökonomischen Einstellungen unterscheiden sich nicht von denen der Landwirte anderer Produktionssysteme.

Für Publikation II wurden Landwirte mit einem Umweltlabel (IP oder BIO) mit Landwirten ohne eine solche Zertifizierung verglichen. Bei der Einstellungsskala zeigten sich bei 10 Einstellungsfaktoren Unterschiede zwischen diesen beiden Gruppen. Lediglich hinsichtlich der Zielstellungen „*Ernährungssicherheit*“ und *„Bereitstellung von Erholungsräumen*“ waren keine Unterschiede feststellbar. Bei allen Einstellungen, bei denen ein Unterschied festgestellt werden konnte, war die Zustimmung zu diesen potentiellen Aufgaben bei den mit Umweltlabel zertifizierten Befragten größer als bei den nicht zertifizierten. Auch bei den Motivationen unterschieden sich diese beiden Gruppen in 10 von 15 Fällen. Die Zustimmung war bei diesen 10 Motivationsfaktoren in der mit Umweltlabel zertifizierten Gruppe höher. Keine Zustimmungsunterschiede konnten bei den folgenden Motivationen, Biodiversitätsförderflächen anzulegen, festgestellt werden: *„weil man dafür Direktzahlungen erhält“, „weil man damit einen Beitrag zum guten Image der Landwirtschaft leistet“, „weil sie aus der Betriebsstruktur heraus Sinn machen“, „weil sie ästhetisch aussehen“* und *„weil dadurch Land genutzt werden kann, das für die Produktion weniger gut geeignet ist“.* Die meisten dieser Motivationen, bei denen keine Unterschiede zwischen Landwirten mit und solchen ohne Umweltlabel festgestellt wurden, können wie auch bei dem Vergleich zwischen Bio- und Nicht-Biobetrieben (Publikation I) den produktionsorientierten bzw. finanziellen Anreizen zugeordnet werden.

Ein weiterer Schwerpunkt war der mögliche Einfluss der Biodiversitätsberatung auf die Einstellungen und Motivationen. Grundsätzlich werden die *„Erhaltung der Bodenfruchtbarkeit*“, die „*Produktion gesunder Lebensmittel*“, ein „*tierfreundlicher Umgang*“ sowie eine „*nachhaltige Nutzung der Ressourcen*“ und eine „*sichere Versorgung der Schweizer Bevölkerung mit Lebensmitteln*“ von allen Landwirten als sehr wichtige Aufgaben eingeschätzt (Publikation II). Hier konnte auch kein Unterschied zwischen beratenen und nicht beratenen Landwirten ermittelt werden. Daraus lässt sich schlussfolgern, dass diese Einstellungen, die ähnlich hohe Zustimmung erhalten haben, einen Teil der Identität der an der Umfrage teilnehmenden Landwirte abbilden. Dazu passt, dass die größten Unterschiede zwischen beratenen und nicht beratenen Landwirten bei jenen Einstellungen festgestellt wurden, die insgesamt die geringste Zustimmung erhalten haben. Dies waren: „*die Bevölkerung wertschätzt die Biodiversitätsförderung durch die Landwirte*“, die „*Förderung der Biodiversität ist für meinen Betrieb wichtig*“ und „*es gibt keinen Widerspruch zwischen Biodiversitätsförderung und Produktion auf meinem Betrieb*“.

Bei den Motivationsfaktoren hingegen waren die Unterschiede durch Beratung bei den Motivationen ausgeprägter, die insgesamt eine größere Zustimmung erreicht hatten. Dazu gehören beispielsweise: *Ich lege Biodiversitätsförderflächen an, … „weil die Landwirtschaft eine Umweltverantwortung trägt*“, „*weil ich einen Beitrag zur Förderung der Biodiversität leisten will*“ und „*weil ich zu einer diversen Agrarlandschaft beitragen möchte*“. Bei den Motivationen war das Zustimmungsniveau aber auch insgesamt nicht so hoch wie bei den Einstellungen, und weist damit ein größeres Verbesserungspotential auf. Dieses Ergebnis kann aber auch ein Hinweis darauf sein, dass die Beratung einen stärkeren Effekt auf die Motivationen hat, denen tendenziell eher zugestimmt wurde. Diese Beziehungen näher zu beleuchten, ist eine Aufgabe für weitere Forschungsvorhaben. Das Ergebnis dieser Untersuchung macht deutlich, dass eine Biodiversitätsberatung die Einstellungen und Motivationen der Landwirte positiv verändern kann. Einschränkend muss angemerkt werden, dass der Stichprobenumfang mit einer vergleichsweise geringen Anzahl von 23 beratenen Landwirten sehr klein war. Die Publikation hat gezeigt, dass es mehrere Einstellungen und Motivationen gab, bei denen durch Beratung eine höhere Zustimmung erzielt werden konnte und dass die Ergebnisse damit verschiedene Handlungsoptionen für Beratungsdienste liefern. Die Tatsache, dass durch Beratung signifikante Unterschiede bei manchen der Motivationen und Einstellungen erzielt wurden, bei andern aber nicht, kann für die Beratung in der Zukunft ein Hinweis sein, sich entweder auf diejenigen zu konzentrieren, bei denen bereits ein Erfolg nachgewiesen werden konnte, oder die Beratungsbemühungen bei denjenigen, bei denen bisher keine Erfolge erzielt werden konnten, zu verbessern. Beide Strategien sind hier zu empfehlen.

Als einschränkender Faktor ist aber zu beachten, dass in den Publikationen I und II nur Betriebe aus dem Talgebiet untersucht wurden und diese sich auch in den Ergebnissen möglicherweise von solchen in höheren Lagen unterscheiden. Zudem entsprach die Stichprobengröße der verschiedenen Produktionssysteme nicht der tatsächlichen Verteilung im Land. Durch die freiwillige Teilnahme an der Befragung besteht auch immer die Gefahr einer möglichen Verzerrung durch die sogenannte Selbstselektion der Teilnehmenden. Darüber hinaus kann auch hier – wie häufig in sozialwissenschaftlichen Untersuchungen – eine Antworttendenz in Richtung sozialer Erwünschtheit nicht ausgeschlossen werden, auch wenn diese dann im Prinzip für alle Einstellungen und Motivationen gelten sollte und sich dort ja durchaus Unterschiede in den Antworten gezeigt haben. Dementsprechend muss man bei der Verallgemeinerung der Ergebnisse vorsichtig sein.

Ein weiterer wichtiger Faktor ist, dass bei dieser Studie allein der Flächenanteil der Biodiversitätsfläche als Maß herangezogen wurde und damit keinerlei Aussage über die Qualität der Biodiversität in den jeweiligen Flächen gemacht werden kann. Dies zeigt einen Ansatzpunkt für weitere Forschungsbemühungen auf, in denen der Zusammenhang zwischen den Einstellungen und der Qualität der Biodiversitätsförderung eine zentrale Rolle spielen sollte. Hierzu muss angemerkt werden, dass die Schweizer Biodiversitätsförderflächen zurzeit in zwei verschiedene Qualitätsstufen (I, II) unterteilt werden. Die Anforderungen aber auch die Biodiversitätsbeiträge für die Qualitätsstufe II sind deutlich höher.

Der ausgezahlte Förderbeitrag setzt sich aus den jeweiligen Flächenanteilen (Qualitätsstufe I und II) auf den Betrieben zusammen. In den eigenen Untersuchungen wurden diese Qualitätsunterschiede allerdings nicht näher betrachtet.

Es hat sich gezeigt, dass die Absicht, Biodiversitätsförderflächen auf Schweizer Landwirtschaftsbetrieben zu implementieren, mit der individuellen Überzeugung der

Landwirte zusammenhängt. Es gibt Hinweise darauf, dass sich die Bereitschaft der Landwirte noch steigern ließe, wenn ein Nachweis über die Wirksamkeit der Biodiversitätsfördermaßnahmen geliefert werden kann. Diese Erkenntnisse liefern auch einige Optionen für landwirtschaftliche Beratungsdienste, die genutzt werden können, um die Bereitschaft der Landwirte für die Teilnahme an solchen Programmen zu erhöhen. Eine Möglichkeit, die Wirksamkeit der Biodiversitätsmaßnahmen zu prüfen, ist der Einsatz von Biodiversitätsbewertungsmethoden. Diese könnten in der Beratung eingesetzt werden, um den Landwirten mögliche Veränderungen durch die Biodiversitätsbewertung aufzuzeigen.

Eine mögliche Form, eine Biodiversitätsbewertung in der Landwirtschaft durchzuführen, wurde in Publikation III näher beleuchtet. In dieser Literaturübersicht wurde ein Überblick über bestehende Biodiversitätsbewertungsmethoden und -ansätze gegeben, die für den Einsatz einer Biodiversitätsbewertung innerhalb von Ökobilanzen geeignet sind. Die dazu aufgestellte Forschungsfrage lautet: Inwieweit sind Biodiversitätsbewertungsmethoden in Ökobilanzen dazu geeignet, die Biodiversitätswirkung der landwirtschaftlichen Produktion zu bewerten?

Die Resultate dieser Publikation zeigen eine Diskrepanz zwischen Methoden, die weltweit anwendbar, aber noch zu grob sind, um landwirtschaftliche Intensitäten zu differenzieren und Methoden, die in der Lage sind, verschiedene landwirtschaftliche Intensitäten zu unterscheiden, dabei aber auf eine bestimmte geographische Region beschränkt bleiben. Daraus lässt sich ableiten, dass zukünftige Forschungsbemühungen zum Ziel haben sollten, diese Lücke zu schließen und weltweit anwendbare Methoden mit regional relevanten Biodiversitätsparametern (Biodiversitätsaspekt, Indikatoren etc.) insoweit weiterzuentwickeln, dass eine Unterscheidung verschiedener landwirtschaftlicher Intensitätsstufen möglich wird.

Die Entwicklung sollte darauf abzielen, dass man die Methoden prinzipiell überall auf der Welt anwenden kann, vor allem wenn sich der ökologische Fußabdruck eines Produktes über verschiedene Länder erstreckt, aber gleichzeitig Biodiversitätsparameter (z. B. Indikatoren) verwendet werden, die in der jeweiligen Region von Relevanz sind. Dabei sollten die Stärken bestehender Methoden gebündelt werden und die Fortschritte genutzt werden, um eine konsequente methodische Weiterentwicklung weiterverfolgen zu können. Die Auswahl wichtiger

Methodenbestandteile, wie der gewählte Biodiversitätsaspekt, die Biodiversitätsindikatoren und die zum Vergleich herangezogene Referenzsituation, sollten zudem immer reflektiert und mit dem aktuellen Wissensstand in der Biodiversitätsforschung abgeglichen werden. Dabei ist zu prüfen, ob die abgeleiteten Schlussfolgerungen mit den ursprünglichen Annahmen übereinstimmen. Ein Beispiel dafür ist die Eignung eines Biodiversitätsindikators für eine Aussage zu einem betrachteten Biodiversitätsaspekt. Dabei sollten der informative Wert und der Auswahlgrund der Methodenbestandteile immer transparent gemacht werden.

Die Grenzen des Methodenvergleichs in Form einer Literaturübersicht bestehen darin, dass es sich um einen rein theoretischen Vergleich handelt und die tatsächliche Anwendbarkeit der verschiedenen Methoden nicht geprüft werden konnte. Hinzu kommt, dass einige der hier verglichenen Methoden bzw. Methodenansätze nicht mit dem Ziel entwickelt wurden, diese für landwirtschaftliche Zwecke einzusetzen. Eine Möglichkeit, diese Einschränkungen zu überwinden, bestünde darin, die Anwendung verschiedener Methoden unter den gleichen Voraussetzungen in einer vergleichenden Fallstudie zu prüfen. Ein solcher Ansatz liefert Erkenntnisse über die Anwendungseigenschaften und kann gleichzeitig einem diagnostischen Zweck dienen.

Diese Empfehlung aus Publikation III, die Anwendung unterschiedlicher Methoden in einer Fallstudie zu testen, wurde in einem ersten Ansatz mit Publikation IV umgesetzt. Allerdings wurden dabei nicht ausschließlich Biodiversitätsbewertungsmethoden für Ökobilanzen eingesetzt, sondern auch eine Ökobilanzmethode und zwei weitere Biodiversitätsbewertungsmethoden, die speziell für den Einsatz auf der landwirtschaftlichen Betriebsebene entwickelt wurden.

Zusätzlich existierten für die ausgewählten fünf Fallstudienbetriebe Artenerhebungsdaten für Pflanzen, Heuschrecken, Schmetterlinge und Vögel. Mit Hilfe der Fallstudie konnte erstens untersucht werden, wie sich verschiedene Biodiversitätsbewertungsmethoden in ihrer Anwendung unterscheiden und welche Vor- und Nachteile sich hinsichtlich ihrer Anwendbarkeit ergeben. Zweitens wurde untersucht, inwieweit sich die Methoden vor dem Hintergrund ihres zugedachten Verwendungszwecks für die Biodiversitätsbewertung landwirtschaftlicher Betriebe eignen.

Der Umstand, dass man Biodiversität aufgrund ihrer Komplexität nicht in ihrer Gesamtheit messen kann (Büchs, 2003), bestätigt sich auch in dieser Fallstudie. Keine der Biodiversitätsbewertungsmethoden ist in der Lage, eine eindeutige, ganzheitliche Aussage über den Biodiversitätszustand der betroffenen Fallstudienbetriebe zu machen. Zudem haben auch die Artenerhebungen gezeigt, dass ein Betrieb beispielsweise von einer Art verhältnismäßig viele Individuen aufweisen kann, von einer anderen Art jedoch im Vergleich mit den anderen Betrieben eher wenige Individuen. Dennoch können die Ergebnisse der verschiedenen Methoden einen Hinweis oder eine Annäherung an den tatsächlichen Biodiversitätszustand oder zumindest an einzelne Aspekte dieses Zustands liefern. Wenn die Einschränkungen der Aussagefähigkeit von Anfang an transparent definiert sind, können die Bewertungsmethoden eine hilfreiche Funktion erfüllen.

Natürlich gibt es in der Betrachtung gerade von Fallstudien gewisse Einschränkungen und so muss z. B. schon aufgrund der geringen Stichprobengröße beachtet werden, dass keine Generalisierungen aus den Ergebnissen abgeleitet werden können (Hays, 2004). Ebenso wenig war der Stichprobenumfang für statistische Analysen hinreichend. Allein deshalb wird empfohlen, eine solche Fallstudie mit einer größeren Anzahl teilnehmender Betriebe durchzuführen. Durch eine bessere Datenverfügbarkeit können in der Zukunft, gerade im Bereich der Ökobilanzmethodik, Zeitersparnisse erzielt werden, die deutlich umfangreichere Stichproben in Fallstudien erlauben. Unabhängig vom gegebenen Stichprobenumfang kann Publikation IV zur Validierung der Methoden für einige Biodiversitätsaspekte herangezogen werden; zudem können die Stärken und Schwächen der verschiedenen Methoden aufgezeigt werden.

Bei dem Vergleich der Methoden mit den Artenerhebungsdaten muss klar sein, dass diese Daten nur den Biodiversitätsaspekt „Artendichte“ darstellen, und zwar nur für die wenigen Indikatorarten (Pflanzen, Heuschrecken, Schmetterlinge und Vögel). Für die beiden Biodiversitätsaspekte „Ökosystemdiversität“ und „genetische Diversität“ gibt es solche Hinweise nicht. Die MVP-Methode, die speziell für den Schweizer Kontext entwickelt wurde, basiert vor allem auf den dort umgesetzten Agrarumweltmaßnahmen und scheint ein recht genaues Bild der Artendichte der Indikatorarten auf den Schweizer Fallstudienbetrieben zu zeigen. Diese Methode ist aber gleichzeitig nicht global einsetzbar und kann deswegen für einen Vergleich von

Betrieben aus unterschiedlichen Regionen der Welt nicht herangezogen werden. Publikation IV hat gezeigt, dass unterschiedliche Aussagen über Biodiversität generiert werden, wenn verschiedene Biodiversitätsbewertungsmethoden auf demselben Betrieb angewandt werden.

Für die Auswahl einer geeigneten Methode sind verschiedene Faktoren entscheidend. So muss beispielsweise die Fragestellung, die durch die Biodiversitätsbewertung beantwortet werden soll, vorher klar definiert sein und es muss überprüft werden, ob sich die Methode für den zugedachten Einsatzzweck überhaupt eignet. Eine Übersicht darüber, bei welcher Fragestellung und Datenlage, unter welcher Voraussetzung, mit welchem Aufwand etc. welche Methode eingesetzt werden kann, gibt es bislang nicht. Dies würde aber sowohl für die Landwirte als auch für die Beratung und weitere Anwender von Biodiversitätsbewertungsmethoden ein sehr hilfreiches Werkzeug darstellen. Alle in der Fallstudie eingesetzten Methoden haben individuelle Stärken und Schwächen. Mit keiner der Methoden gelingt eine deutlich treffgenauere Abschätzung der Biodiversität.

5 Ausblick

Eine standortspezifische Biodiversitätsberatung durch gut geschulte Berater kann eine ergänzende Strategie zu den allgemeinen, nicht regionsspezifischen, aber bereits etablierten Biodiversitätsförderprogrammen sein. Die Biodiversitätsberater detektieren dabei gemeinsam mit den Landwirten das Biodiversitätspotential auf den Betrieben, definieren die Schutzziele und legen passende Maßnahmen fest.

Biodiversitätsbewertungsmethoden können den Beratern helfen, die Landwirte als Partner für die Biodiversitätsförderung zu gewinnen; insbesondere dann, wenn Biodiversitätsveränderungen auf den Betrieben rasch sichtbar werden. Landwirte wollen wirkungsvoll agieren und die Erfolge ihrer Bemühungen sehen können, um sich als Biodiversitätsförderer verstehen und darstellen zu können. Fremdeinschätzung und Anerkennung aus der Gesellschaft sind wichtige Motivatoren für die Landwirte.

Ein weiterer Ansatz kann darin bestehen, das Wissen und das ehrenamtliche Engagement kundiger Naturschützer vor Ort einzubeziehen, um Biodiversität mit den Landwirten im partnerschaftlichen Modus zu fördern. Dabei können Vorurteile und Differenzen beseitigt und neue Brücken geschlagen werden. Die Offizialberatung kann dabei moderierend wirken.

Die auf den Betrieben eingesetzten Maßnahmen sollten Ertragsausfälle auf den Produktionsflächen möglichst vermeiden. Deshalb ist zunächst eine kostenneutrale Biodiversitätsförderung auf ertragsarmen Teilflächen oder angrenzend zu den Kulturflächen anzustreben. Diese Vorgehensweise ist besonders in der Schweiz wichtig, wo schon jetzt etwa die Hälfte der Nahrungsmittel importiert wird. Führt die Biodiversitätsförderung zu deutlichen Ertragsverlusten, wird die Abhängigkeit von Importen gesteigert, ein Sachverhalt der potentiell mit gesteigerten Umweltlasten und einem Biodiversitätsschwund an Produktionsstandorten außerhalb der Schweiz verbunden sein kann. Erhöhten Umweltleistungen in der Schweiz stünden dann gesteigerte Umweltlasten an externen Produktionsstandorten gegenüber. Hier findet die Idee, die gesamte Wertschöpfungskette von landwirtschaftlichen Produkten mit Hilfe von Ökobilanzen global zu verfolgen, um Umweltlasten und -leistungen auch an anderen Orten in der Welt zu quantifizieren, ihre Grenzen. Im Gegensatz zu den abiotischen Wirkungskategorien in Ökobilanzen kann die Biodiversitätserhaltung und

-steigerung derzeit über den betrieblichen/regionalen Rahmen hinaus nicht zufriedenstellend erfasst und umgesetzt werden. Es wird in absehbarer Zeit wohl keine Methode geben, die in der Lage ist, verschiedene landwirtschaftliche Produktivitätsintensitäten hinsichtlich der Wirkungskategorie Biodiversität zu erfassen, und die gleichzeitig in verschiedenen Teilen der Welt angewandt werden kann.

Auch in Zukunft wird man in der Schweiz für signifikante Biodiversitätsleistungen auf Direktzahlungen oft nicht verzichten können. Direktzahlungen sind die Voraussetzung jedweder nicht kostenneutralen Strategie. Aus den Ergebnissen der Landwirtsbefragungen wissen wir, dass ein monetärer Ausgleich für die Landwirte dann eine wichtige, wenn nicht die wichtigste Motivation darstellt. In dieser Einstellung unterscheiden sich Biolandwirte oder Landwirte mit einem anderen Umweltlabel (IP) nicht von den konventionellen Kollegen.

Um durch eine gezielte Beratung noch bessere Wirkung zu erzielen, sollten noch zwei weitere Strategieelemente verfolgt werden. Zum einen müsste man die Beratungsbemühungen in Bezug auf die Einstellungen und Motivationen intensivieren, bei denen die Ergebnisse dieser Arbeit schon einen Beratungseffekt aufgezeigt haben. Zum anderen sollten gerade bei den Schlüsselfaktoren, die bislang keine Veränderungen durch Beratung gezeigt haben, die Beratungsstrategien gezielt intensiviert und zu höherer Effektivität gesteigert werden.

Als Grundlage weiterer Forschung und einer stetigen Neu- und Weiterentwicklung der Methoden empfiehlt es sich, sowohl für den nationalen als auch für den internationalen Gebrauch, eine Katalogisierung aller bestehenden Methoden zu entwickeln, inklusive Darstellung der einzelnen Methodenparameter. Damit kann ein Überblick über die bestehende Methodenvielfalt gegeben werden, die es zukünftigen Anwendern ermöglicht, eine geeignete Methode für ihre Fragestellungen und die Datenlage vor Ort auszuwählen. Gleichzeitig kann auf diese Weise klar definiert werden, welche Aussagen überhaupt anhand der Methodik getroffen werden können. Dabei sollte deutlich auf die Stärken, aber auch auf Schwächen und Grenzen der vorhandenen Methoden hingewiesen werden.

Dafür ist Transparenz notwendig, hinsichtlich

- der angestrebten Verwendungszwecke,

- der von den Methoden abgedeckten Biodiversitätsaspekte,
- der Eignung der verwendeten Biodiversitätsindikatoren,
- des geographischen Anwendungsraumes und
- der benötigten Datenmenge und Datenqualität.

Gezielte Forschungsförderung ist nötig. So sind beispielsweise Techniken des „*Precision Farming*“ zum sensorischen Bildverstehen, z. B. für die Detektion zu fördernder Wildkrautflora, bislang nicht hinreichend entwickelt. Mit der Strategie der *„Public Private Partnerships“* ließen sich durch Start-up-Unternehmen hier vermutlich rasch in der Praxis anwendbare Techniken entwickeln. Somit könnte künftig die zu schützende bzw. zu fördernde Ackerwildflora effizient erkannt und eine an die Bedarfe der Pflanzen angepasste, sensorgestützte, teilflächenspezifische Bewirtschaftung ermöglicht werden.

Die vorliegenden publizierten Forschungsergebnisse machen deutlich, dass es ein großes Potential für die biologische, aber auch für die konventionelle Landwirtschaft und ihre Bauern gibt, um einen signifikanten und wirkungsvollen Beitrag zur Erhaltung der noch vorhandenen Biodiversität zu leisten. Eine konsequente, lukrative und praxisorientierte Forschungsförderung ist essenziell, um Biodiversität zu erhalten und zu steigern und damit die bereits verursachten Biodiversitätsverluste zumindest teilweise zu kompensieren.

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Motivations for swiss lowland farmers to conserve biodiversity: Identifying factors to predict proportions of implemented ecological compensation areas

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1. Introduction

The global loss of biodiversity is one of the major environmental threats of our time (Rockstrom et al., 2009) and agriculture is one of the main drivers of the global biodiversity decline (Benton, 2007; Kleijn et al., 2009; Pereira et al., 2010). Agricultural landscapes were formerly rich in biodiversity, which has been attributed to mosaic style landscapes and low-intensity production systems that provided a wide variety of interlinked habitats (Edwards et al., 1999). Economic pressure has led to declining species richness as agricultural production has become more intensive (Robinson and Sutherland, 2002), with an associated reduction of habitat-providing landscape elements (Billeter et al., 2008). Despite the objective of promoting biodiversity, which has been included in Swiss agricultural policy since 1990, many of the threatened species continue to decline (Lachat et al., 2010).Due to the large proportion of farmed land, the behavior and the decision-making of farmers in respect to conservation and sustainability issues have an extraordinary influence on biodiversity (Lokhorst et al., 2011; Stoeckli et al., 2017). Rands et al. (2010) argue that, if biodiversity decline is to be halted, biodiversity must be viewed as a public good and this view must be integrated into policy. Encouraging farmers to preserve or enhance biodiversity is often achieved through agri-environmental schemes that provide financial rewards to enhance biodiversity (Burton and Paragahawewa, 2011). Most agri-environmental schemes are based around principles in which subsidies, or compensation payments, are linked to the farmer's compliance with a set of environmental measures, with subsidies paid in exchange for proof of ecological performance (PEP) (Kleijn and Sutherland, 2003).

Several studies have questioned the effectiveness of subsidy based agri-environmental schemes and have shown that they do not always lead to achieving biodiversity goals in agricultural landscapes (Lütz and Bastian, 2002; Bengtsson et al., 2005; Schenk et al., 2007; Billeter et al., 2008; Henle et al., 2008; Burton and Paragahawewa, 2011; Greiner and Gregg, 2011; Batary et al., 2015). In general though, Carvalheiro et al. (2013) concluded that European agri-environmental schemes have at least slowed the loss of farmland biodiversity, while Batary et al. (2015) suggested they may have led to moderate increases. Given that a major driver of biodiversity loss in agricultural landscapes is due to the loss of habitat-providing landscape elements (Billeter et al., 2008), it is reasonable to conclude that provision of habitats can contribute to biodiversity conservation. The primary basis for agri-environmental schemes in Switzerland is to provide habitats, in the form of low-input or semi-natural habitats, which are called 'ecological compensation areas' (ECA). In an evaluation of the effectiveness of ECAs, Herzog and Walter (2005) found that they contribute to biodiversity with measurable benefits, Knop et al. (2006) found they have positive effects on biodiversity, and Birrer et al. (2014) concluded that the area of ECA is the best available proxy for biodiversity at farm level.

Schroeder et al. (2015) noted that both the European Court of Auditors (2011) and the European Commission have demanded further improvements to agri-environmental schemes to increase their efficiency and effectiveness. Billeter et al. (2008) summarized the opinion of many experts that further biodiversity decline can only be stopped with major changes in policy which have to be supported through better developed agri-ecological farm practices and enhanced scientific knowledge.

Batary et al. (2015), in their meta-analysis of literature on the effectiveness of agri-environmental schemes, found that motivations for engagement go beyond utilitarian motives, such as payment rate and ease of fit within existing farm practice, to also include a range of factors including farmers' attitudes and circumstances. de Snoo et al. (2010) pointed out that behavioral aspects need more scientific attention, while Ahnström et al. (2013) called for more understanding of farmers' attitudes and beliefs regarding biodiversity in agricultural landscapes. Organic farmers have been found to collectively possess exceptionally positive attitudes towards biodiversity (Läpple, 2012), to be more favourable towards ecological measures (Schader et al., 2008), and to have an average of 30% higher species richness than

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conventional farms (Bengtsson et al., 2005). Conversely, Läpple (2012) found that conventional farmers were the most profit oriented and least environmentally aware group of farmers. These findings suggest that the motivations of organic and non-organic farmers are different, so a comparison of organic and non-organic farmers is relevant to this study.

The aim of this contribution is to address these challenges and examine the influence of motivational factors on the implementation of habitat-providing landscape elements on farms. Given their prominence in the literature on motivations of farmer behavior, particular attention will be given to direct payments as part of an agri-environmental scheme, a secondary aim of this study is to investigate whether there are differences between organic and non-organic farmers in the strengths of the identified motivational factors. We use the case of the Swiss lowlands, which provides an example of a subsidy payment agri-environmental scheme and can therefore contribute to the knowledge of how farmers might be motivated to engage with biodiversity conservation to make such schemes become more effective.

1.1. Contextual background

Direct payments are usually necessary for the financial survival of Swiss farms, and are paid to Swiss farmers on the condition that they establish and maintain ECAs, with prescribed quality standards (BLW, 2013), on at least seven percent of their farm's agricultural production area (Lanz and Lehmann, 2012). Almost all Swiss farms meet this national requirement and are therefore entitled to receive subsidies, which include direct payments for ECAs. No farmers have the obligation to implement ecological measures on more than the required 7% of their farmland, although many farmers exceed this amount voluntarily and receive an incremental increase in direct payments to compensate for the extra area that is allocated as ECAs.

Farmers in Switzerland may participate in one of two main labelling schemes that require them to implement additional measures to enhance biodiversity, which are often used to improve the quality of their ECAs. The 6348 organic farms represent 12.1% of all Swiss farms (BFS, 2017b) and receive an extra subsidy for organic production along with higher product prices. The 20 000 IP SUISSE farms represent 38.3% of all Swiss farms (IPSUISSE, 2017) and have to carry out ecological measures from a predefined list if they are to be accredited as integrated production (IP) farms (Birrer et al., 2014). IP SUISSE farmers also get higher product prices but receive no specific subsidies from the government for practicing IP. All farmers can decide for themselves which of the prescribed measures they want to implement as part of their ECAs, and both organic and IP SUISSE farmers can choose which additional quality measures to implement so as to qualify for the respective label (Jenny et al., 2013). There are approximately 26 000 (49.6%) of the 52 263 Swiss farms (BFS, 2017a) that do not belong to any voluntary label organization and which are called proof of ecological performance (PEP) farms (BLW, 2013).

1.2. Motivations of farmers

One potential policy instrument to force environmental behavior by farmers is simply to legislate. However, Bartel and Barclay (2011) concluded that a policy based on understanding the attitudes towards government, environmental problems, environmental laws and regulations, and farm management behaviors is more appropriate than further regulations. Pretty and Smith (2004) point out that intended behavior tends to change, and biodiversity conservation may be compromised, when regulations cease to be enforced. Similarly, Barnes et al. (2013) found that forced behavior can lead to negative attitudes towards environmental issues and thus can unintentionally create an even higher need of regulations. It appears that legislation alone is not enough, but Aviron et al. (2009) pointed out that the intended outcomes from regulation can be enhanced if regulation is supplemented by additional mechanisms such as financial incentives.

Financial incentives, in the form of subsidies paid for desirable behavior, are a further, and potentially powerful, policy instrument to motivate environmental behavior by farmers (Aviron et al., 2009). The assumption behind this strategy is that farmers are primarily motivated by profit maximisation, so financial incentives are the best way to motivate them to provide biodiversity benefits for society (Hanley et al., 2012). This assumption receives empirical support by Cary and Wilkinson (1997) who found that perceived profitability was the most important factor influencing the implementation of conservation practices and exceeded the individual farmer's conservation orientation. On the other hand, Frey and Oberholzer-Gee (1997) and Burton and Paragahawewa (2011) have questioned whether extrinsic financial incentives might even weaken, rather than strengthen, intrinsic motivation. Pretty and Smith (2004), while acknowledging their effectiveness, argue that the combination of regulation and financial incentives is also not sufficient to achieve long term biodiversity targets but can be complemented by the individual farmers being committed to, and identifying with, the objectives of the measures. Wilson and Hart (2000, p. 2161) suggest that 'the financial imperative for participation does not necessarily exclude an often equally important environmental concern' and de Snoo et al. (2010) demanded that behavioral aspects need more scientific attention from researchers interested in environmental attitudes and behavior among farmers. De Snoo et al. (2013: 66) put it simply that "We should aim to place farmland biodiversity 'in the hands and minds of farmers'". Lokhorst et al. (2011) point out that intrinsic motivations that are related to the intention to perform voluntary actions, which in this case, can also be understood as engaging with prescribed actions, are associated with the concepts of self-identity and personal norms.

Hewitt, in the first edition of his book in 1991, defined the social psychological concept of identity as one's personal location within social life, in which identities are meanings we attribute to ourselves and are learned from others' expectations of how we should behave (Stedman, 2002). People have multiple roles, which means they hold multiple identities that are associated with different aspects of their lives, so we tend to organize Identities hierarchically in accordance with their importance or salience (Stryker and Burke, 2000). Role-person merger occurs when a role becomes critical to a person's self-definition (Stedman, 2002) and it is reasonable to assume that the role of 'farmer' is sufficiently important that it will be dominant in a farmer's identity.

Burton (2004) was among the first scholars to examine the influence of farmer identity, which led to farmer identity taking a place within the existing, theorizing literature in rural sociology. McGuire et al. (2015) describe farmer identity as the social psychological framework to help better understand how farmers perform and see themselves, while Fielding et al., 2008 sum up a growing body of research to show that identity is an essential predictor of behavioral intentions. In this sense, Groth and Curtis (2017) stated that the occupational identity of farmers, which is an example of role-person merger (Stedman, 2002), influences their land use and land management. Burton (2004) pointed out the dominance of the productivist paradigm in the European agricultural landscape, which led him to conclude that very little was then known about the farmer's perspective of 'production' and 'productivism'. Home et al. (2014) in their study of Swiss farmers, proposed that farmer identity lies on a continuum ranging from wholly a productivist through to wholly a conservationist. Productivist farmers organize their farms so that the requirements for direct payments are fulfilled with minimum disruption to production, while conservationist farmers organize their farms so that production causes minimum disruption to the environment.

van Dijk et al. (2015) found a significant relationhip between Dutch farmers' identity and their intention to participate in agri-environmental schemes, but added that the relationship between self-identity and farmers' intention is strongly influenced by other factors such as social pressure and their relationship with the farmer collectives who administer the schemes. Morris and Potter (1995) investigated the motivations of English farmers to participate in agri-environmental schemes and found wide variations in the level of commitment and sympathy with the wider objectives of the schemes; ranging from resistant non-adopters to active adopters. The identity of Austrian farmers influenced the decisions on the farms with production-oriented farmers having fewer plant species on their farms than traditional or innovative farmers (Schmitzberger et al., 2005). Home et al. (2014), in a qualitative study, found that Swiss farmers' degree of identification with the dual roles of farmers as producers and conservationists was a major influence on their intention to implement agri-environmental measures.

Conservation measures on farms are more likely to be implemented if the farmers perceive them to be relevant (Burton et al., 2008) and to be easily integrated into the day to day running of the farm (Jahrl et al., 2012). However, de Snoo et al. (2010) reported no correlation between socio-psychological differences of Dutch farmers and the environmental impacts of nature conservation measures and so could not attribute actual behavior to intrinsic motivations. Similarly, Jahrl et al. (2012) found only weak correlations between intrinsic motivation and the quantity and quality of ecological compensation areas that had actually been implemented. A further factor that has been found to influence uptake of biodiversity conservation measures are social barriers, (Rodriguez et al., 2009), with the opinions of neighbouring farmers being an important influence on farmers' decisions on whether to join agri-environmental schemes (Siebert et al., 2006; Defrancesco et al., 2008; Hynes and Garvey, 2009; Home et al., 2014). On the other hand, Schroeder et al. (2015) found that neighbors' opinions only play a minor role.

This review of existing literature has identified several contradictions in both the relative influence of extrinsic and intrinsic motivations for implementation of ecological compensation areas on farms. It appears that primary research is necessary to evaluate whether beliefs and motivational factors lead to environmental outcomes. Understanding how beliefs and motivations can be influenced and how they can lead to actions, will allow the development of strategies to increase biodiversity on farms through specifically improved advice.

2. Methods

2.1. Sample

The sample consisted of a group of 303 farmers who were randomly selected from a database of Swiss farms. The selected farms were located in the German-speaking lowland and hill production zones of the Swiss Central Plateau below 800 m a.s.l. in the cantons: Aargau, Bern, Lucerne, Solothurn and Zurich. All farms were similar sized, mixed production farms. Organic farmers were oversampled to enable a comparison of organic and non-organic farmers by collecting data from similarly sized samples. The sample was stratified so that the three main production systems in Switzerland: organic (151 sent), IP Suisse (76 sent) and PEP (76 sent) were included in the survey. The questionnaire was sent by post, with a pre-addressed and pre-paid return envelope, in autumn 2015.

Responses were received from 133 farmers (response rate 43,9%). The response rates from the sub-samples according to production system were organic (n = 60; 40%), IP Suisse (n = 36; 47%) and PEP (n = 37; 49%). The respondents reported farm sizes with an average of 24.6 ha (SD 4.3) and an average proportion of arable crops of 39.1% (SD 17.1), which closely corresponds to the national lowland averages (BFS, 2017a). The proportion of ECAS on the responding farms ranged from 7% to 70%, with a median proportion of 12.0 Ha. Only six of the responding farms reported the minimum required 7% of ECAs, while eight farms reported greater than 25% of their farms as being managed as ECAs. No significant difference ($p = 0.252$) was found in the proportion of implemented ECA between organic (mean = 13.95%) and non-organic farms (mean = 15.92%).

2.2. Survey design

In the social sciences, a combination of several empirical indicators (known as items) in a composite measure is referred to as a scale (DeVellis, 2003). Based on the review of relevant literature and qualitative interviews reported in Home et al. (2014), we developed a 10 item scale to evaluate whether respondents believe that specific outcomes belong to the tasks of farmers. Items were formulated to reflect a perceived responsibility to provide a good or service for a relevant third party, such as the next generation or the general population. Five further questions were included about farmers' general beliefs with regard to biodiversity and nature (listed in Table 1). Three of these items: ‘sustainable use of natural resources’, ‘maintaining and promoting diverse cultural landscapes’, and ‘providing and maintaining recreation areas’, are explicitly mentioned in the goals of the current Swiss agricultural policy: AP 14–17 (BLW, 2012), because of their contribution to the general well-being of the Swiss population, but they are not directly related to productive farming. A farmer's identification with these goals can therefore be considered to be in line with the social norms held by the Swiss population. Responses to these survey items were on a five point Likert scale from 1: ‘strongly disagree’ to 5: ‘strongly agree’.

Based on the review of relevant literature and the same qualitative interviews, we created a 15-item scale (listed in Table 2) to evaluate motivations. Five items refer to a general philosophy of the farmer about the internal motivations for implementing an ECA on their farm. Five further items refer to motivations in terms of the practical outcomes of implementing the ECA in relation to fitting in with the operational structures and day to day farm running, which includes, in the case of the ‘direct payments’ item, whether the ECA contributes to income. The remaining items explicitly refer to the existing public and to future generations, including two items: the ‘diverse landscape’ and ‘aesthetically beautiful’ items, which directly refer to the stated goals of the current Swiss agricultural policy: AP 14–17 (BLW, 2012). Responses to these survey items were on a five point Likert scale from 1: ‘strongly disagree’ to 5: ‘strongly agree’. Demographic data were collected including farm type and farming system. Furthermore, the farmers were asked about the proportion of their farms on which ECA had been implemented.

2.3. Analysis

Lokhorst et al. (2011) have criticized the observed tendency of researchers to measure only farmers' attitudes, rather than other constructs that might influence behavior. On the other hand Kollmuss and Agyeman (2002), pointed out that developing a model that incorporates all the factors behind pro-environmental behavior might not be usable, and suggest the value of a simplified model to aid in clarifying and categorising such factors. One theory that appears promising to order and understand the data is the Theory of Planned Behavior (TPB) (Ajzen, 1991), which suggests that an intention to engage in a particular behavior is based on three factors: attitude toward the behavior, social norms, and perceived behavioral control. The TPB has frequently been used in the context of farmer decision-making, such as for pesticide use (Heong and Escalada, 1999), agro-forestry practices (Zubair and Garforth, 2006) and implementation of ecological compensation areas in the Netherlands (van Dijk et al., 2016) and England (Schroeder et al., 2015). The TPB was amended to include actual behavioral controls (Ajzen, 2002) and provides a useful framework for considering whether belief and motivational factors can be considered to be perceived or actual behavioral controls. According to the TPB, perceived behavioral controls can be expected to correlate with associated belief items, while actual behavioral controls can be expected to be independent of belief items. Specifically, rather than to tease out which factors can be classified as attitudes, subjective norms or perceived behavioral controls, we use the amended TPB to evaluate whether direct payments are, as a perceived behavioral control, simply one of several motivating factors, or whether they should indeed be considered separately (see Fig. 1).

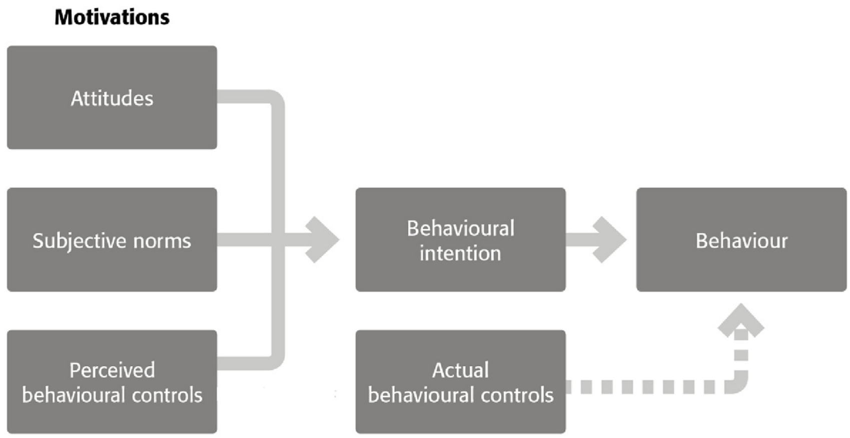

Fig. 1. Theory of planned behavior (adapted from Ajzen, 2002).

The survey responses were analyzed using three statistical procedures: all of which were calculated using SPSS Version 17.0. Spearman's rank-order correlations were conducted to identify correlations between scale items and the proportion of implemented ECA on the respondents' farms. Mann-Whitney U Tests were used to identify group differences in the responses to the motivations scale. In both cases, non-parametric tests were selected because of the uncertainty of normal distribution from the five point scales used in this study. The third procedure was a principal components analysis (PCA) of the motivations scale.

PCA is a mathematical procedure that transforms a number of (possibly) correlated variables into a (smaller) number of uncorrelated variables, called principal components, to reveal underlying explanations for patterns in a set of data (Hotelling, 1933). The aim of PCA is to select an optimal number of components, which is defined as the minimum number of components that accounts for the maximum possible variance in the correlation matrix. The accumulated percentage of explained variance gives an indication of the goodness of fit of the component solution (Lorenzo-Seva, 2013). Varimax is an algorithm that attempts to rotate the principal components so that individual variables tend to be associated with just one component (Fabrigar et al., 1999). The factor loadings, also called component loadings in PCA, are the correlation coefficients between the variables and principal components (Fabrigar et al., 1999), which represent how much a factor explains a variable in the PCA. The highest loadings signify which factor affects that item the most. Cases in which two component loadings for an item are of similar magnitude allow the item to be considered to be similarly affected by both components (Abdi and Williams, 2010).

3. Results and discussion

3.1. Correlations between scale items and implemented ECAs

The mean responses to the questions about beliefs are shown in Fig. 2. For presentation reasons, the variable names have been used while the detailed information (including standard deviations and number of respondents) is shown in Table 1.

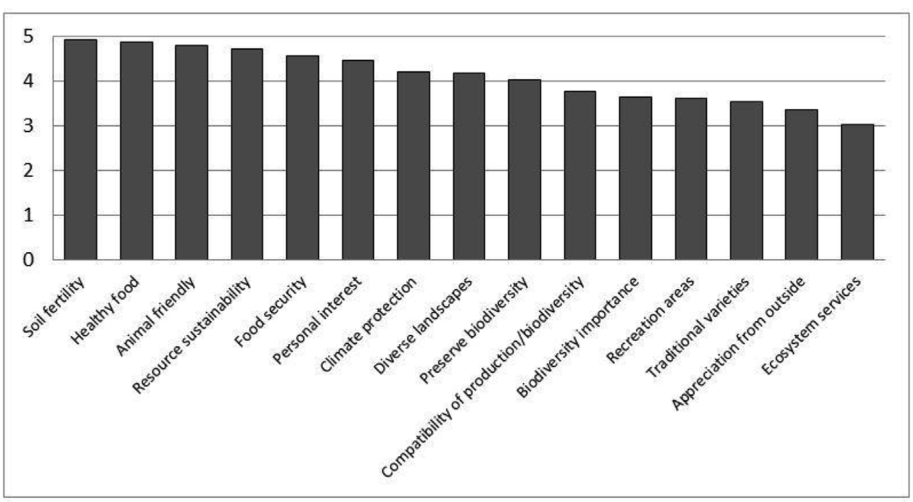

Fig. 2. Mean responses of farmers to 'beliefs' variables (1 = Strongly Disagree to 5 = Strongly Agree).

The finding that soil fertility is considered by the responding farmers to be the most important of their tasks is consistent with the findings of Casagrande et al. (2016) who found that soil conservation was the most important motivation to adopt conservation practices in agriculture.

The mean responses to the questions about motivations are shown in Fig. 3. For presentation reasons, the variable names have been used while the detailed information (including standard deviations and number of respondents) is shown in Table 2.

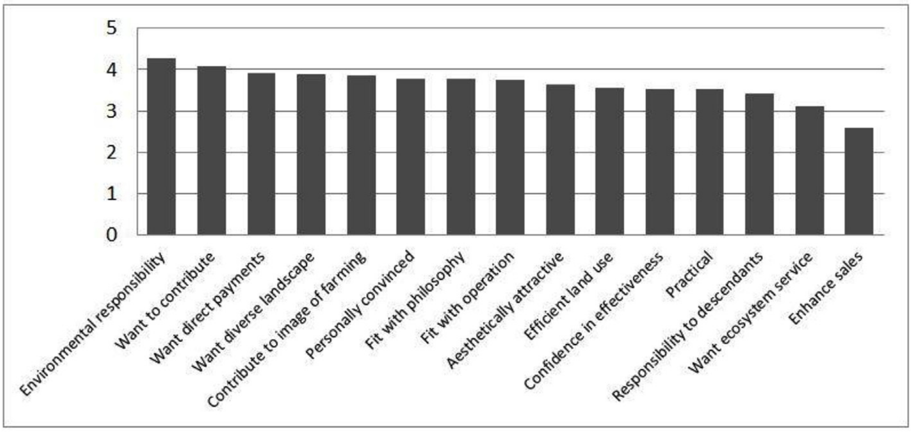

Fig. 3. Mean responses of farmers to motivations variables (1 = Strongly Disagree to 5 = Strongly Agree).

There was less agreement with the motivations items than with the beliefs items. In particular, the motivation to implement ECA to enhance sales found little agreement. Apparently, the responding farmers do not perceive ecological performance to add value to the products, which implies a lack of trust that consumers will support their ecological preferences with purchases. This finding echoes the results of Home et al. (2014) who found that Swiss farmers' sense of fairness was irritated when consumers request ecologically sensitive farming but purchase cheaper imports, which have less stringent ecological requirements. The highest agreement within the motivations items was given for 'environmental responsibility', indicating that the farmers have attributed more importance to a sense of responsibility (an inner motivation) than, for example, to the external motivation through direct payments.

Nevertheless, direct payments, with the third-largest agreement, seem to play an important role, which is underlined by the 'direct payments' item correlating most strongly with implemented ECAs (shown in Table 2). These results agree with the findings of Cary and Wilkinson (1997) and Aviron et al. (2009) who concluded that financial incentives are among, or even the, most important factor to motivate environmental behavior of farmers; but contrast the findings of van Dijk et al. (2015) who found that perceived profitability had no significant influence on farmers intention to implement conservation measures on their farms. Van Dijk et al. (2015) explanation for this discrepancy was that farmers may over-emphasise financial reasoning when implementing environmental measures so as to present themselves as rational business people, as had also been found by Sutherland (2011). The results of this study suggest that the opposite might be the case in that the payments appear to be among the most important factors, which means that the stated environmental responsibility might be the over-emphasised justification. A correlation analysis was used to identify which of the 'belief' items significantly correlated with the proportion of effective ECA on the farms. The results are shown in Table 1.

Table 1
Correlation between belief variables and proportion of ecological compensation area.

Variable name	Scale items (respondents' agreement with these statements)	Mean	SD	Spearman's rho	p-value	N
Soil fertility	Important task is maintaining the soil fertility for the next generation	4.92	0.29	0.00	0.982	120
Healthy food	Important task is production of healthy food	4.88	0.48	−0.15	0.104	119
Animal friendly	Important task is animal friendly livestock farming	4.80	0.45	0.03	0.745	120
Resource sustainability	Important task is sustainable use of natural resources soil, water and air	4.71	0.61	0.06	0.520	120
Food security	Important task is ensuring a secure supply of food for the Swiss population	4.56	0.80	−0.08	0.413	120
Personal interest	I have a personal stake in the animal and plant worlds on my farm	4.47	0.75	0.03	0.748	118
Climate protection	Important task is climate protection	4.21	0.87	0.06	0.511	118
Diverse landscapes	Important task is maintaining and promoting diverse cultural landscapes	4.19	0.87	0.11	0.233	118
Preserve biodiversity	Important task is preservation and promotion of biodiversity	4.03	0.97	**0.21**	0.025	120
Compatibility of production/ biodiversity	For me, there is no contradiction between biodiversity conservation and production	3.76	1.19	0.14	0.123	118
Biodiversity importance	Promotion of biodiversity is important for my farm	3.64	1.06	**0.32**	< .001	119
Recreation areas	Important task is provision and maintenance recreation areas	3.61	1.09	**0.24**	0.010	118
Traditional varieties	Important task is conservation of traditional varieties and breeds	3.54	1.07	0.17	0.062	118
Appreciation from outside	I feel the general population appreciate our work with regard to biodiversity promotion	3.37	1.03	**0.19**	0.042	118
Ecosystem services	In my experience, ecosystem services, such as pest control or pollination have been improved by installing ecological compensation areas	3.03	1.08	**0.21**	0.031	107

Numbers in bold show significant correlations.

Five of the 15 given belief items correlated positively with the proportion of implemented ECA on the farms, with Spearman's rho ranging from 0.19 to 0.32. Farmers who reported stronger agreement with these beliefs have a higher proportion of ecological compensation area on their farms than their colleagues who reported less agreement. The five items that correlated positively with the proportion of implemented ECA are those, with the exceptions of 'compatibility of production/biodiversity' and 'traditional varieties', with the lowest agreement by the farmers (means ranging from 3.03 to 4.03) and with the highest variation (standard deviations ranging from 0.97 to 1.09). With the exceptions of 'compatibility of production/biodiversity' and 'traditional varieties', all of the belief items that did not significantly correlate with the proportion of ECA are those with most agreement by the farmers (means ranging from 4.92 to 4.19), and with least variation (standard deviations ranging from 0.29 to 0.87).

Organic farmers reported significantly stronger agreement with seven of the items from the beliefs scale, which, with the exceptions of the 'healthy food' (p = 0.013) and 'traditional varieties' (p = 0.001) items, were the items related to biodiversity and conservation. These included 'personal interest' (p = 0.004), 'climate protection' (p = 0.003), 'compatibility of production/biodiversity' (p = 0.024), and 'biodiversity importance' (p = 0.007). One item that is clearly related to biodiversity and conservation is the 'preserve biodiversity' item, for which no differences between the groups could be distinguished (p = 0.397). There were no significant differences between the groups with regard to the production oriented items including the soil fertility (p = 0.159), resource sustainability (p = 0.131), or food security items (p = 0.173).

A correlation analysis was also used in order to identify which of the 'motivation items' significantly correlated with the proportion of ECA on the farms (Table 2).

(p = 0.006), '*ECA, because they fit into the philosophy of the production of our farm*' (p = 0.001), '*ECA because I have confidence that the measures make sense and are effective*' (p = 0.015), and '*ECA because I believe agriculture has an environmental responsibility*' (p = 0.02). There were no significant differences (p > 0.05) between organic and non-organic farmers in responses to any of the motivation items related to financial gain.

3.2. Direct payments as part of an agri-environmental scheme

The principal components analysis revealed a Cronbach's alpha of 0.932, which suggests adequate internal reliability (Tavakol and Dennick, 2011). The Kaiser-Meyer-Olkin index (0.903) and the Bartlett Test of Sphericity (p < 0.001) confirm sampling adequacy and suggest that the data are appropriate for a Principal Component Analysis respectively (Fabrigar et al., 1999). Two components with Eigenvalues greater than 1 were extracted. The rotated (Varimax with Kaiser normalization) solution converged in 3 iterations. The factor loadings (greater than 0.300) are displayed in Table 3.

Two components emerged in the Rotated Component Matrix. The first component explains 46.4% of the variance and contains items related to the farmers' conservation and sustainability orientation, and can be described as 'intrinsic motivations'. These components appear to correspond with the theory of planned behavior and can be assigned to attitudes, subjective norms, and perceived behavioral controls.

The second component explained a further 17.3% of the variance and contains items related to 'economic motivations', which includes financial gain. The dominant items in the 'economic motivations' component, which loaded strongly only against this component, were *'ECA, because I receive direct payments for them'* and '*ECA, because I can use land, which is less suitable for production'*. Both these items are clearly related to financial gain and production, and refer to factors which influence the behavior that appear to be, at best, only loosely related to farmers' attitudes.

Table 2
Correlation between motivation variables and proportion of ecological compensation area.

Variable name	Scale items (respondents' agreement with these statements)	Mean	SD	Spearman's rho	p-value	N
Environmental responsibility	ECA because I believe agriculture has an environmental responsibility	4.26	0.91	0.07	0.468	116
Want to contribute	ECA, because I want to contribute to the conservation and promotion of biodiversity	4.08	1.09	**0.28****	0.003	115
Want direct payments	ECA, because I receive direct payments for them	3.92	1.19	**0.30****	0.001	116
Want diverse landscape	ECA, because they contribute to a diverse agricultural landscape	3.88	1.18	**0.19***	0.045	115
Contribute to image of farming	ECA, because I want to contribute to the good image of agriculture	3.85	1.16	**0.19***	0.046	115
Personally convinced	ECA, because I am internally convinced	3.77	1.15	**0.22***	0.021	114
Fit with philosophy	ECA, because they fit with the philosophy of production on our farm	3.77	1.28	**0.28****	0.003	113
Fit with operation	ECA, because they make sense from the operational structure of the farm	3.75	1.19	**0.26***	0.006	115
Aesthetically attractive	ECA, because they are aesthetically beautiful to look at	3.62	1.18	0.11*	0.234	114
Efficient land use	ECA, because I can use land that is less suitable for production	3.54	1.28	**0.21***	0.024	116
Confidence in effectiveness	ECA, because I have confidence that the measures make sense and are effective	3.52	1.23	0.15	0.103	115
Practical	ECA, because they are practical and do not interfere with production	3.52	1.27	**0.32****	0.001	115
Responsibility to descendants	ECA, because I feel morally committed to future generations	3.40	1.22	0.09	0.367	116
Want ecosystem service	ECA, because they give me an ecological service (e.g., pest control)	3.11	1.22	**0.30****	0.002	113
Enhance sales	ECA, because they have a positive effect on my product sales	2.60	1.27	**0.31****	0.001	116

Numbers in bold show significant correlations.
*Correlation is significant at the 0.05 level (2-tailed).
**Correlation is significant at the 0.01 level (2-tailed).

With the exception of four items: *'I believe agriculture has an environmental responsibility'*, '*ECA`s because they are aesthetically beautiful to look at'*, *'I have confidence that the measures make sense and are effective'* and *'I feel morally committed to future generations'*, all of the motivation variables correlated positively with the proportion of implemented ECA, with Spearman's rho ranging from 0.19 to 0.32. This result supports their selection as motivational factors and gives confidence in the validity of the scale.

Differences between organic and not organic farmers in terms of motivation for implementation of ECAs were only found in five of the motivational factors: all of which were related to an environmental orientation. These were '*ECA, because I am convinced of it internally*'

Four items loaded against both components. One of these: '*ECA because I want to contribute to a good image of agriculture*', might reflect that having a profitable farm also contributes to the image of farmers. A second item: '*ECA, because they are aesthetically beautiful to look at*', also loaded against both components but considerably more strongly against the intrinsic motivations component. Two other items that loaded against both components: '*ECA, because they make sense from the operational structure*' and '*ECA, because they have a positive effect on my*

Table 3
Rotated Component Matrix for the motivation variables.

Rotated Component Matrix[a]	Component	
	Intrinsic motivations	Economic (incl. financial gain)
ECA, because I am convinced of it internally	.885	
ECA, because they fit into the philosophy of the production of our farm	.867	
ECA because I have confidence that the measures make sense and are effective	.866	
ECA, because I want to contribute to the conservation and promotion of biodiversity	.830	
ECA, because they contribute to a diverse agricultural landscape	.807	
ECA, because I feel morally committed to future generations	.759	
ECA because I believe agriculture has an environmental responsibility	.723	
ECA, because they give me an environmental benefit (e.g., pest control)	.706	
ECA because they are practical and do not interfere with production	.671	
ECA, because they make sense from the operational structure	.610	.440
ECA, because they are aesthetically beautiful to look at	.587	.427
ECA, because they have a positive effect on my product sales	.490	.482
ECA, because I receive direct payments for them		.825
ECA, because I can use land, which is less suitable for production		.800
ECA, because I want to contribute to a good image of agriculture	.560	.594

Extraction Method: Principal Component Analysis.
Rotation Method: Varimax with Kaiser Normalization.
[a] Rotation converged 8 iterations.

product sales', can be seen as intrinsic motivations, but also clearly contribute to economic gain.

A further correlation analysis revealed no significant correlation (P values greater than 0.05) between the 'direct payments' item and the key belief variables (those that were found to correlate with the proportion of ECA), which suggests that the 'direct payments' motivation factor is independent of these key beliefs. The 'efficient land use' item, which also loaded solely against the economic motivations component, correlated significantly with two of the key belief items: 'recreation areas' ($p = 0.003$) and 'preserve biodiversity' ($p = 0.047$).

4. Conclusions

These findings, that there exists a conservation component and an economic component, correspond with the results of Home et al. (2014), who proposed that Swiss farmer's identity is located on a continuum between a production and a conservation orientation. The results of the PCA, combined with the results of the correlation analysis of the direct payments item and the key belief items, suggest that direct payments belong to a different type of motivation and maybe an actual rather than a perceived behavior control (see Fig. 4).

The finding that there were significant ($p < 0.05$) differences between organic and non-organic farmers in environmentally oriented motivation items but no significant differences in responses to any of the motivation items related to financial gain supports the conclusions of Läpple (2012) that conventional farmers are less conservation oriented, but contradicts her conclusion that organic farmers are less profit oriented. This is further reflected in the finding that the organic farmers did not have a significantly ($p = 0.252$) larger proportion of ECAs on their farms, which suggests that the production orientation of the organic farmers, along with their need to retain economic viability, is sufficiently strong to mediate their ecological beliefs. In other words, their beliefs are expressed in their choice of production system, but they remain productive farmers with their motivations also including the economic component that they have in common with non-organic farmers.

All of the surveyed farmers reported strong agreement that the tasks of farmers include maintaining the soil fertility for the next generation; production of healthy food; farming in a way that is animal friendly; using natural resources sustainably; and ensuring a secure supply of food for the Swiss population. Although these responses might be a result of socially desirable responding, the observation that these items received stronger agreement than the remaining items is likely to be

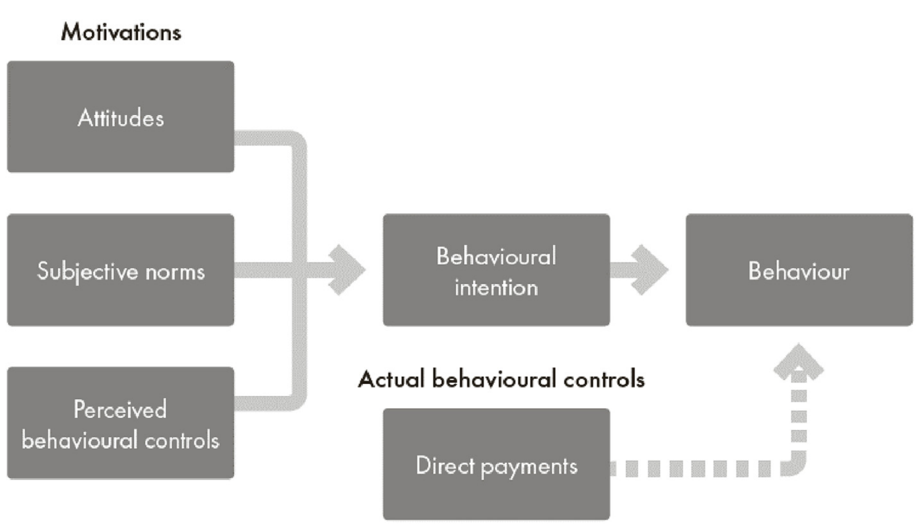

Fig. 4. Theory of planned behavior with the illustrated role of direct payments as an actual behavioral control (adapted from Ajzen, 2002).

reliable because it is likely that any response bias was applied to all questions. For example, '*provision of recreation*' and '*appreciation from outside*' can also be considered to be socially desirable, but received less agreement. A similar reasoning allows an assumption of reliability of the correlation analysis between beliefs, motivations and the proportion of ECA on the farms, which identified a range of beliefs and motivations that correlate with the implementation of ECA. An oversampling of organic farmers, for whom ecological responsibility is part of the underlying philosophy, might have biased these results and it will be the task of future research to investigate this further.

With regard to the motivation factor labelled 'direct payments', the Spearman's rho statistic in the correlation analysis ranged from 0.172 to 0.438 and the item loaded against the 'economic' factor in the PCA. These results suggest that the 'direct payments' motivation factor is independent of these key beliefs, while strongly correlating with the implemented proportion of ecological compensation areas. Based on these findings, we have classified 'direct payments' as an actual, rather than a perceived, behavioral control. This finding provides empirical support for the arguments of Home et al. (2014), Schroeder et al. (2015), and van Dijk et al. (2016) who have each suggested that direct payment systems should be complemented by social and/or psychological motivation strategies. Regardless of an individual farmer's belief, financial survival is a fundamental requirement of being a farmer, and only very few farmers in Switzerland can survive financially without direct payments. This finding was reflected in the result that organic farmers did not respond significantly differently to the motivations items that were related to production, despite reporting stronger alignment with the items that were ecological in nature. Despite a stronger ecological orientation, which is reflected in their choice of production method, organic farmers remain primarily producers and must also be economically viable if they are to survive.

Switzerland provides an example of a subsidy payment based agri-environmental scheme and the results of this study contribute to understanding how the beliefs and motivations of farmers might be used to increase the effectiveness of such schemes. However, all of the participating farms were located in the Swiss lowlands, so there may be some differences to farms at higher elevations, which tend to be more pasture based, or to farms in other national contexts with differences in how agri-environmental schemes are administered. Despite this limitation, the evaluation of beliefs and motivations can be understood as reflecting the mentality of what it means to be a farmer, and it is plausible that this worldview, and these results, will apply across a broad range of contexts. However, further study will be required to confirm their generalizability and thereby confirm their use as tools for those interested in motivating farmers to engage with on-farm biodiversity.

The proportions of the respondents from the three main production systems: organic, IPSuisse, and PEP, in the sample do not accurately reflect the proportions in Switzerland, and there is a potential bias due to self-selection of respondents, in this case farmers, which is a common characteristic of social research that relies on the voluntary participation. Therefore, caution should be taken when making generalizations about Swiss farmers based on the descriptive statistics. A further limitation is that the study only focused on the proportion of ECA on the farm. It was beyond the scope of the study to include the quality of implemented ECAs in the analysis, which would be evaluated by the impact of ECAs on biodiversity, so further study would be needed to evaluate the relationships between beliefs and quality. A reasonable hypothesis is that knowledge and expertise in implementation along with cultural differences between farmers in different cantons will play an important role. Despite these limitations, the sample size was adequate for statistical analysis, and the inter-item correlations can be considered to be reliable because it is reasonable to assume that any socially desirable response bias was constant across responses.

This study has shown that intention to implement ecological compensation areas on Swiss farms was related to the individual beliefs held by the participating farmers. Furthermore, the study allows the conclusion that demonstration of the effectiveness of agri-environmental measures, in terms of both conservation and production, will increase farmers' willingness to implement them, which provides some options for farm advisory services. For example, biodiversity assessment methods could be used as a tool for advisors to demonstrate the effects of agri-environmental measures, such as implementation of ecological compensation areas, to farmers.

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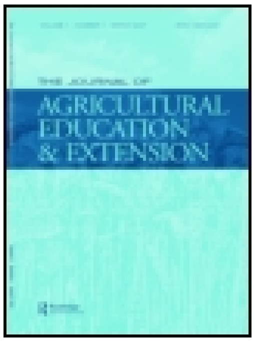

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The influence of on-farm advice on beliefs and motivations for Swiss lowland farmers to implement ecological compensation areas on their farms

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ABSTRACT

Purpose: Farmers hold a key to reaching biodiversity targets, but will only carry out this service to society if they are sufficiently motivated to do so. The aim of this study is to evaluate the potential of on-farm advice as a tool for motivating farmers to take action to preserve or even to enhance biodiversity on their farms.

Design/methodology/approach: To address this aim, we surveyed 133 farmers (response rate 43.9%), of whom 23 had received on-farm advice about farmland biodiversity conservation over a period of six years.

Findings: The results showed that key beliefs and motivations were positively influenced by farmer advisory services. Farmers who had received advice agreed significantly more strongly in the compatibility of biodiversity conservation and production; that biodiversity is important; and that nature conservation on farms is appreciated by society.

Practical Implications: These results allow the conclusion that on-farm biodiversity advice might be a useful way of positively influencing the beliefs and enhancing motivations of farmers to contribute to biodiversity conservation on their farms.

Theoretical implications: Although several papers have examined the influence of agricultural extension on farmer behaviour, the results of this study contribute to explaining some of the contradictions in the literature about the effectiveness of advisory services. Furthermore, this study addresses [Batary, P., L. V. Dicks, D. Kleijn, and W. J. Sutherland. 2015. "The Role of Agri-Environment Schemes in Conservation and Environmental Management." *Conservation Biology* 29: 1006–1016.] challenge that there has been insufficient research on the link between farmer advice and the effectiveness of agri-environmental schemes.

Originality/value: This paper is among the first to address these research gaps and is the first to examine the influence of advice on farmer conservation behaviour in the Swiss context.

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1. Introduction

Declining worldwide biodiversity is among the major environmental threats faced by humanity (Rockstrom et al. 2009). According to the Convention on Biological Diversity (CBD) at the World Summit on Sustainable Development held in Johannesburg in 2002, the loss of species was intended to be significantly reduced by 2010 (Eisermann 2003). As a result, 87% of CBD signatories have developed national biodiversity strategies and action plans (Butchart et al. 2010). Agriculture plays an important role within the Swiss Biodiversity Strategy (BAFU 2011) and the associated action plan calls for concrete measures for agriculture (BAFU 2017).

However, at a conference in Nagoya in 2010 it was reported that none of the contracting parties had reached this goal (BAFU 2011). Although the objective of promoting biodiversity has been included in Swiss agricultural policy since 1990, many threatened species in Switzerland continue to decline (Lachat et al. 2010).

Species loss and the decline of biodiversity are largely driven by the loss of habitat-providing landscape elements (Billeter et al. 2008). This is particularly relevant to agricultural areas because habitats are seldom adequately preserved in intensively used agricultural land (Hole et al. 2005), and intensive agriculture is common in Europe due to economic pressure on farmers (Robinson and Sutherland 2002). Semi-natural habitats that could potentially be created in agricultural landscapes can contribute to meeting biodiversity targets. Rosin (2013) argues that a move towards such landscapes, which he calls food utopias, is essential if we are to achieve a just and sustainable food system.

However, the tendency in rural areas in developed countries is that, rather than providing areas of biodiversity conservation, intensification is increasing and the description of conservation oriented farming landscapes as being utopian (Rosin 2013) has the connotation that they are a worthy, but perhaps unattainable, target. If agricultural landscapes are the front line of biodiversity conservation, the managers of agricultural land, namely farmers, can play a key role in achieving biodiversity conservation targets.

The role of farmers has been investigated within the discussion about farmer identity, which is defined by McGuire et al. (2015) as the social psychological framework which helps to explain the performance and the self-image of farmers. As one of the first, Burton (2004) investigated the influence of farmer identity and emphasized that the farmers' point of view on 'production' and 'productivism' was not sufficiently well known. Furthermore, Burton and Wilson (2006) noted that, despite an increasing demand for farmers' self-conception to contain at least a portion of 'conservationist' orientation, production-oriented identities still dominate. This means that farmers tend to be productivist, with a broadly accepted notion that a good farmer is one who produces commodities, with more production seen as being better than less production. McGuire, Morton, and Cast (2013) state that, although a large number of farmers already have a conservationist-identity, their conservation goals must be activated in order to counterbalance the productivist orientation and achieve 'production-conservation meanings' which they see as demanded from society.

There is evidence that the conservation goals of farmers are becoming aligned with their production goals, with grassroots and cooperative efforts, such as 'paying for ecosystem services', aiming to reflect societal demand for greening of agricultural practice, with acceptance of the agricultural value of ecosystem services (Wynne-Jones 2013). This acceptance has led to a change in which 'the environmental and socio-cultural benefits of farming are portrayed as positive by-products of "multifunctional-agriculture", to present "ecosystem goods and services" as desirable commodities in their own right' (Wynne-Jones 2013, 77). In Switzerland, the ability to pass on a healthy farm to the next generation is included in the concept of what makes a good farmer (Home et al. 2014). Stock (2007, 88) concludes 'that good farming implies a moral and reflexive concern for the environment and an explicit concern for the health and wellbeing of their customers and people in general.'

Despite this change in attitude, biodiversity still retains many of the characteristics of a public good, which includes non-rivalry and non-excludability (Ferraro and Kiss 2002; Heal 2003), and is therefore prone to freeriders and consequent market failure, so government intervention is usually required for the efficient provision of public goods (Osemeobo 1993). Such provision is being undertaken on many fronts, such as nature reserves and national parks, but privately owned farms cover a considerable amount of territory and they are becoming increasingly intensively managed (Jongman 2002). Governments therefore take measures to encourage farmers to contribute to the provision of biodiversity.

One governmental measure would be simply to legislate for particular behaviours, but this approach encounters resistance because farmer autonomy is also included in the farmer self-identity (Stock and Forney 2014). There is a real risk that rules will simply be rejected, so a more promising approach is for governments to be convincing. de Snoo et al. (2013, 66) suggest that we should aim to 'place farmland biodiversity in the hands and minds of farmers', but it remains unclear how best to motivate farmers to adopt biodiversity conservation into their hearts and minds.

The aim of this study is to address this research gap by evaluating whether, in the context of Switzerland, beliefs and motivational factors can potentially lead to environmental outcomes. A further aim is to evaluate whether beliefs and motivations can be influenced by biodiversity-specific on-farm advice. Understanding how beliefs and motivations can be influenced and how they can lead to actions, will allow the development of strategies to increase biodiversity on farms through specifically improved advice. Organic farmers form a special case in that they receive advice as part of their organic certification. A final aim of this study is to evaluate whether organic farmers agree more or less strongly than non-organic farmers, with beliefs and motivational factors that have been found to correlate with environmental outcomes.

2. Literature review

Measures to encourage farmers to preserve or enhance biodiversity are often variations of agri-environmental schemes in which subsidies, or compensation payments, provide farmers a financial incentive for enhancing biodiversity (Burton and Paragahawewa 2011). Most agri-environmental schemes require farmers to provide proof of ecological performance (PEP), which is achieved by the farmer demonstrating compliance with a set of environmental standards, for which subsidies are paid (Kleijn and Sutherland 2003). However, Home et al. (2014) argue that the identity of farmers is production oriented and question whether financial motivations, which are often seen as being paid to not produce, is the most suitable strategy. Burton and Schwarz (2013) argue that result-oriented agri-environmental schemes, which provide incentives to farmers to gain knowledge and improve their skills, have the potential to motivate long-term behavioural change. Rands et al. (2010, 1301) argue that simply demonstrating the value of biodiversity is insufficient to motivate behavioural change 'unless supporting public policies are in place that either reward positive individual actions or penalize harm'. Reimer, Thompson, and Prokopy (2012) suggest that a combination of both financial instruments and psycho-social motivations might better reflect the rural reality than the reliance on purely financial incentives.

Batary et al. (2015) pointed out that managing land in order to achieve environmental outcomes requires different skills and knowledge than those gained in typical farmer education, which suggests the value of specialist on-farm advisory services. For example Noe, Halberg, and Reddersen (2005) found that information was most effective in influencing farmers' perceptions and awareness of wildlife when delivered in dialogue with a biologist. Agri-environmental schemes are often amended for scientific and/or political reasons, and their dynamic nature further supports the need for specialized advice to farmers so they can adapt to changes (Smallshire, Robertson, and Thompson 2004). Chevillat et al. (2012) proposed that the willingness for the implementation of measures to enhance biodiversity could be raised considerably for farmers who receive on–farm biodiversity advice from biodiversity experts at whole farm level, while Winter, Mills, and Wragg (2000) identified a direct correlation between farmers who had received direct on-farm advice and those with the best biodiversity conservation plans. Smallshire, Robertson, and Thompson (2004) claim that direct on-farm advice by well-trained advisors is the most effective, nevertheless expensive, way of providing information to farmers.

Schroeder, Chaplin, and Isselstein (2015), on the other hand, found that the opinions of farm advisors had little influence on farmers' decisions to participate in agri-environmental schemes, which they explain by noting that farm advisors held a relatively neutral opinion. Sutherland et al. (2013) pointed out that advisors may be incentivized to help farmers maximize their agri-environmental grants rather than enabling actions with the most environmental benefit: particularly in cases where the advisors represent services perceived to be one-sidedly oriented towards the short-term production function of the farms. Ironically, advisory services that are perceived to be impartial or one-sidedly oriented towards the short-term production function of the farms are those that are most trusted by farmers and are therefore more likely to influence farmer behaviour in the short term (Sutherland et al. 2013). Chevillat et al. (2017) found that Swiss farmers who received targeted advice about how to both enhance biodiversity on their farms, but also how to maximize the direct payments they receive for implementing biodiversity conservation measures, implemented a significantly higher quantity of ecological compensation area (ECAs) on their farms. ECAs are parts of agricultural land which have been less intensively managed or set aside in order to create habitats for plants and wildlife (BAFU 2014). Given the apparent contradictions between findings about the effectiveness of advisory services, it is not surprising that Batary et al. (2015) comment that there has been insufficient research on the link between farmer advice and the effectiveness of agri-environmental schemes.

2.1. Contextual background

As in most European countries, the Swiss Confederation pays subsidies to Swiss farmers to provide and maintain semi-natural habitats on their farms (BLW 2013) with the requirement that at least seven percent of each farm's agricultural area must be reserved as ECA (Lanz and Lehmann 2012). Practically all Swiss farms meet this requirement and so are eligible to claim the subsidies that are needed for financial survival. Farms which meet the minimum environmental requirements to receive subsidies, and do not participate in additional voluntary labelling schemes, are called proof of ecological performance (PEP) (BLW 2013). Alternatively, Swiss farmers can choose to additionally participate in one of two labelling schemes.

One is organic farming, which, as part of their organic certification, must meet stricter conditions than the minimum environmental requirements for direct payments. Organic farmers receive an extra subsidy for organic production along with higher product prices. Organic farmers have been found to perceive biodiversity to be an integral part of their farms (Rahmann 2002). In Switzerland, organic farmers are strongly encouraged to seek professional advice, including both production oriented and biodiversity advice, and must attend a compulsory two day training course, as part of their certification process (FiBL 2017). Indeed, organic farmers have been found to be more favourable towards ecological measures (Schader et al. 2008) and to have an average of 30% higher species richness than conventional farms (Bengtsson, Ahnström, and Weibull 2005; Tuck et al. 2014).

IP-Suisse is a private label in which integrated production (IP) farmers have to implement ecological measures, from a pre-defined list, to gain accreditation (Birrer et al. 2014). IP systems strive to produce high quality food through the use of natural resources and regulating mechanisms, and thereby allow reduced application of synthetic inputs (Boller et al. 2004). IP-Suisse farmers do not receive an extra subsidy for IP, but do receive higher product prices.

All farmers can choose which measures to implement as part of the minimum requirement, and can also implement additional measures for which they receive additional subsidies, with the amount of payment depending on the measure. Both IP-Suisse farmers and organic farmers also have freedom to choose which additional measures to implement for them to qualify for their respective label (Jenny et al. 2013).

3. Methods

3.1. Survey design and sampling

In this study we use two scales, with 15 items each, which we call the 'farmer beliefs scale' and the 'farmer conservation motivations scale'. The 'farmer beliefs scale' consists of items to evaluate whether respondents believe that specific outcomes belong to the tasks of farmers and to evaluate their general beliefs with regard to biodiversity and nature. The 'farmer conservation motivations scale' includes items that refer to motivations in terms of the practical outcomes of implementing the ECA in the operational structures and day to day farm running; items that explicitly refer to the existing public and to future generations; and items that refer to the stated goals of the current Swiss agricultural policy: AP 14–17 (BLW 2012). Responses to the items on both scales were on a five point Likert scale from 1: 'strongly disagree' to 5: 'strongly agree'. Demographic data were collected including farm type and farming system.

The sample consisted of a group of 25 farmers who, between 2008 and 2015, had received regular and ongoing on-farm advice about biodiversity as part of a the project called 'Scoring with biodiversity—farmers enrich nature' (Birrer et al. 2009). The sample was supplemented with 25 farmers who participated in the 'Scoring with biodiversity' project but had not received on-farm advice; and a representative sample of 142 organic farmers, 68 IP-Suisse farmers and 68 PEP farmers. It was beyond our control whether farms who were not part of the 'scoring with biodiversity' project had received biodiversity advice and we cannot be sure that they did not. However, if it happened that some farmers had received such advice, we could expect between group differences to be smaller, rather than greater.

The selection criteria for the supplementary farms were that they should be similar in size, production, and location to the initial 25 farms. Farms were selected which were located in the lowland and hill production zones of the Swiss Central Plateau below 800 m a.s.l. and were all mixed production farms. In the case of organic farmers, Bio Suisse, the umbrella body for organic farmers in Switzerland, provided the random sample of supplementary farms from their database, which had been filtered to meet the selection criteria. The remaining supplementary farms were extracted from the Swiss telephone directory, which could be searched according to location and production type, but not by farm size. Participants were randomly selected from the extracted list and a screening question was added to the questionnaire so that any excessively large or small farms could have been excluded. Organic farmers were oversampled so that sample sizes for between-group comparisons were similar. The questionnaire was sent by post, with a pre-addressed and pre-paid return envelope, in autumn 2015. Respondents were given the option of completing the questionnaire electronically if they preferred.

Responses were received from 23 of the 25 farmers who had received advice within the 'Scoring with biodiversity' project (response rate 92%) and from 110 farmers in the remaining sample (response rate 36%). The response rates from the whole sample according to production system were organic (n = 60; 40%), IP-Suisse (n = 36; 47%) and PEP (n = 37; 49%). The respondents reported farm sizes with an average of 24.6 ha (SD 4.3) and an average proportion of arable crops of 39.1% (SD 17.1), which closely corresponds to the national lowland averages (BFS 2017).

3.2. Analysis

Mann-Whitney U Tests were used to identify group differences in the responses to the beliefs and farmer conservation motivations scales. Non-parametric tests were selected because of the uncertainty of normal distribution from the five point scales used in this study. All results were calculated using SPSS Version 17.0.

4. Results and discussion

In an initial step, responses to both scales were compared between the sample of farmers who had participated in the 'Scoring with biodiversity' project but had not received on-farm advice and the randomly selected sample of farmers. No significant between-group differences could be identified, which suggests that the samples could be merged into a 'non-advised' farmer group. Furthermore, responses to both scales were compared between organic and IP-Suisse farmers. The Mann-Whitney U Tests revealed significant differences in three items from the farmer beliefs scale: '*climate protection (item B7, see*

Table 1. Differences in beliefs between not advised and advised farmer; and labelled and non- labelled (PEP) farmers.

Nr.	Item	Label	Mean not advised *N* = 110	Mean advised. *N* = 23	*P*-value	Mean non-labelled (PEP) *N* = 37	Mean labelled *N* = 96	*P*-value
B1	Important task is maintaining the soil fertility for the next generation	*Soil fertility*	4.91	4.96	0.575	4.86	4.95	**0.053**
B2	Important task is production of healthy food	*Healthy food*	4.90	5.00	0.156	4.86	4.88	–
B3	Important task is animal friendly livestock farming	*Animal friendly*	4.80	4.83	0.605	4.72	4.83	**0.073**
B4	Important task is sustainable use of natural resources soil, water and air	*Resource sustainability*	4.67	4.87	0.173	4.47	4.80	**0.017**
B5	Important task is ensuring a secure supply of food for the Swiss population	*Food security*	4.64	4.43	0.288	4.69	4.51	0.316
B6	I have a personal stake in the animal and plant worlds on my farm	*Personal interest*	4.43	4.65	0.245	4.11	4.61	**0.003**
B7	Important task is climate protection	*Climate protection*	4.25	3.95	0.146	4.00	4.28	–
B8	Important task is maintaining and promoting diverse cultural landscapes	*Diverse landscapes*	4.17	4.26	0.727	3.94	4.29	**0.064**
B9	Important task is preservation and promotion of biodiversity	*Preserve biodiversity*	3.98	4.30	0.177	3.72	4.15	**0.082**
B10	For me, there is no contradiction between biodiversity conservation and production	*Compatibility of production and biodiversity*	3.62	4.39	**0.003**	3.32	3.93	**0.025**
B11	Promotion of biodiversity is important for my farm	*Biodiversity importance*	3.53	4.05	**0.054**	3.08	3.85	**0.001**
B12	Important task is provision and maintenance recreation areas	*Recreation areas*	3.55	3.82	0.366	3.49	3.65	0.542
B13	Important task is conservation of traditional varieties and breeds	*Traditional varieties*	3.50	3.64	0.723	2.91	3.77	**<0.001**
B14	I feel the general population appreciate our work with regard to biodiversity promotion	*Appreciation from outside*	3.24	3.91	**0.004**	3.08	3.48	**0.034**
B15	In my experience, ecosystem services, such as pest control or pollination have been improved by installing ecological compensation areas	*Ecosystem services*	2.95	3.32	0.190	2.58	3.20	–

Tables 1 and 2 *for item numbering)*' ($p = 0.007$; organic mean = 4.42, IP mean = 4.06), '*healthy food (B2)*' ($p = 0.007$; organic mean = 4.93, IP mean = 4.80), and '*ecosystem services (B15)*' ($p = 0.037$; organic mean = 3.37, IP mean = 2.94); but revealed no significant differences in responses to the farmer conservation motivations scale. Based on this result, the organic farmers and the IP-Suisse farmers were grouped, and called 'labelled' farmers, for comparison with the responses from the PEP farmers.

4.1. The farmers' beliefs scale

The Cronbach's alpha of the 'farmer beliefs scale' was 0.848, which is well above the accepted threshold of 0.70. All items gave a lower alpha value when removed except for the item 'Food security', for which the scale alpha value improved to 0.864 when this item was removed. Observation of the inter-item correlations also revealed a low correlation with other items, which suggests that this item might belong to another dimension. It was anyway not significantly different between groups (this result will be discussed later), so played no further role in the analysis or discussion. It remains a challenge for future research to further investigate the dimensionality of these scales, and such an analysis is beyond the scope of this paper. So we have left the item in the analysis so that it may be considered in such future study.

To investigate whether beliefs and motivations can be influenced by giving on-farm advice to farmers, we examined differences between the advised and non-advised farmers. Furthermore we compared responses from organic, IP-Suisse and PEP farmers to the beliefs and farmers conservation motivations scales.

The results of the tests for the farmer beliefs scale are shown in Table 1. No comparisons were made between labelled and non-labelled farmers for the three 'belief' variables for which significant differences were found in responses from organic and IP-Suisse farmers because the labelled farmers cannot be considered to be a homogenous group with regard to these variables.

4.2. Belief differences between advised and non-advised farmers

The results revealed ($p < 0.1$) differences in responses to three items between the advised/non-advised groups, two of which were significant at the 0.05 level. Strong agreement from all farmers was received for several of the items for which no between-group differences for advised and non-advised farmers were identified, which suggests that these might be fundamental beliefs that contribute to the identity of being a farmer. These include strong beliefs that it is the task of farmers to maintain soil fertility, to produce healthy food, to farm in an animal friendly way, to use natural resources sustainably, and to ensure national food security.

However, significant differences between the advised and non-advised farmers were found for the items: '*compatibility of production and biodiversity (B10)*' ($p = 0.003$), '*biodiversity importance (B11)*' ($p = 0.054$) and '*appreciation from outside (B14)*' ($p = 0.004$), with the farmers who had received advice agreeing more strongly with these belief variables than the non-advised farmers. These strong beliefs were either pre-existing and led to the farmers accepting advice or, as Dweck (2008) suggested might be the case, were able to be influenced by the advisors: despite the resilience of an individual's

Table 2. Differences in motivations between not advised and advised farmer; and labelled and non-labelled (PEP) farmers.

Nr.	Item	Label	Mean not advised **N = 110**	Mean advised **N = 23**	*P*-value	Mean non-labelled PEP **N = 37**	Mean label **N = 96**	*P*-value
M1	ECA because I believe agriculture has an environmental responsibility	*Environmental responsibility*	4.18	4.64	**0.025**	3.94	4.38	**0.018**
M2	ECA, because I want to contribute to the conservation and promotion of biodiversity	*Want to contribute*	4.00	4.45	**0.060**	3.78	4.20	**0.053**
M3	ECA, because I receive direct payments for them	*Want direct payments*	3.86	4.22	0.199	4.03	3.88	0.396
M4	ECA, because they contribute to a diverse agricultural landscape	*Want diverse landscape*	3.74	4.55	**0.001**	3.44	4.06	**0.005**
M5	ECA, because I want to contribute to the good image of agriculture	*Contribute to image of farming*	3.79	4.14	0.197	3.58	3.96	0.150
M6	ECA, because I am internally convinced	*Personally convinced*	3.69	4.14	**0.059**	3.17	4.01	**<0.001**
M7	ECA, because they fit with the philosophy of production on our farm	*Fit with philosophy*	3.67	4.25	**0.042**	3.06	4.06	**<0.001**
M8	ECA, because they make sense from the operational structure of the farm	*Fit with operation*	3.69	4.05	0.248	3.61	3.91	0.700
M9	ECA, because they are aesthetically beautiful to look at	*Aesthetically attractive*	3.60	3.73	0.696	3.39	3.72	0.298
M10	ECA, because I can use land that is less suitable for production	*Efficient land use*	3.57	3.43	0.588	3.69	3.48	0.281
M11	ECA, because I have confidence that the measures make sense and are effective	*Confidence in effectiveness*	3.45	3.86	0.151	3.00	3.73	**0.004**
M12	ECA, because they are practical and do not interfere with production	*Practical*	3.44	3.95	**0.077**	3.14	3.67	**0.065**
M13	ECA, because I feel morally committed to future generations	*Responsibility to descendants*	3.33	3.74	0.186	3.00	3.57	**0.020**
M14	ECA, because they give me an ecological service (e.g. pest control)	*ECA, because they give me an ecological service (e.g. pest control)*	3.10	3.19	0.755	2.66	3.29	**0.007**
M15	ECA, because they have a positive effect on my product sales	*ECA, because they have a positive effect on my product sales*	2.57	2.73	0.582	2.31	2.71	**0.096**

beliefs (Bandura 1989). A possible explanation for the relatively few differences between the advised and non-advised (fewer than between labelled and non-labelled) is that beliefs are normally resilient and people are reluctant to change them (Bandura 1989).

4.3. Belief differences between labelled and non- labelled farmers

The results revealed differences in responses to ten items between the labelled and non-labelled groups ($p < 0.1$), six of which were significant at the 0.05 level. Only for the items '*Food security (B5)*' and '*recreation areas' (B12)* no significant differences could be found. The differences in beliefs between labelled and non-labelled' groups of farmers appears to be more pronounced than the differences found between advised and non-advised farmers. These stronger beliefs could be part of the reason that the farmer converted to organic production or joined the IP-Suisse certification scheme.

The finding that labelled farmers have stronger positive environmental beliefs is in agreement with Schader et al. (2008) who found that organic farmers held a more positive attitude than non-organic farmers towards agri-environmental measures on their farms and with Rahmann (2002) who stated that biodiversity conservation is an integral part of the identity of organic farmers. We have found that these beliefs are also evident for IP-Suisse farmers.

Organic farming explicitly entails beliefs such as production of healthy food, and the importance and compatibility of biodiversity and ecosystem services, as part of its core values, which are conveyed to farmers who convert to organically certified production during a compulsory two day training course as part of the conversion process (FiBL 2017). The basic philosophy of IP-Suisse includes that IP-Suisse farmers should actively encourage biodiversity on their farms, participate in further education, and that their production should be environmentally friendly: thereby contributing to the production of healthy food (IP-Suisse 2017). Therefore farmers who believe more strongly in these belief items may have already converted to organic or IP-Suisse production, while farmers who believe less strongly in these items may have felt discouraged to join a labelling scheme.

4.4. The farmer conservation motivations scale

In the farmer conservation motivations scale, all items gave a lower alpha when removed except for the item: direct payments. However the increase in alpha was marginal (from 0.932 to 0.935); the item correlated strongly with two other items when inter-item correlations were calculated; and no significant between group differences were found. These findings suggest that the scale might be multidimensional, which would be a worthwhile topic for future research, so the item was presented in the results but not discussed further.

The results of the tests for the farmer conservation motivations scale are shown in Table 2.

4.5. Motivations differences between advised and non-advised farmers

The results revealed differences in responses to six items ($p < 0.1$), three of which were significant at the 0.05 level, between the advised/non-advised groups. The group of farmers who had received biodiversity advice reported stronger agreement than their colleagues who did not receive biodiversity advice that they were '*personally convinced (M6)*', that agriculture bears an '*environmental responsibility (M1)*' and that promoting biodiversity '*fits to their philosophy (M7)*' and is '*practical (M12)*'. Furthermore, the advised farmers agreed more strongly that it is important to '*contribute to biodiversity (M2)*', and to have '*diverse landscapes (M4)*'. Chevillat et al. (2017) found that farmers who had received targeted advice had larger proportions of ECAs on their farms, which suggests that the advice may have a positive effect on biodiversity protection on the farm. However, there may also have been a selection effect of farmers who were already more motivated for these kind of measures and therefore were open to participating in the on-farm advice scheme. A lack of a before and after comparison of the beliefs does not allow definitive conclusions to be drawn about an effect of the advice. Farmers who had received on-farm advice agreed more strongly with scale items that indicate favourable attitudes towards biodiversity: namely the belief that agriculture has an environmental responsibility and a willingness to contribute to the conservation and promotion of biodiversity. The finding supports the findings of Winter, Mills, and Wragg (2000), Smallshire, Robertson, and Thompson (2004) and Chevillat et al. (2017) who consider on-farm advice to be an effective and meaningful measure to promote biodiversity on farms. The differences associated with receiving farmer advice also appear to exist, in some cases, at the level of beliefs, which may be the result of the advice: a premise that is supported by Dweck's (2008) assertion that beliefs can be changed by targeted interventions. At first glance, this finding appears to contradict the conclusions of Schroeder, Chaplin, and Isselstein (2015) who found that the opinions of farm advisors had little influence on farmers' decisions to participate in agri-environmental schemes. However the farm advisors in Schroeder, Chaplin, and Isselstein's (2015) study tended to give advice about profitability and productivity and held neutral opinions on the topic of biodiversity, while the advisors in this study gave farmers specific and specialized biodiversity advice. Farmers who had received this specialized biodiversity advice gave measurably different responses than those who had not received such advice.

With two exceptions, the 'farmer conservation motivations scale' items that had received stronger agreement from all farmers were those for which significant differences between advised and non-advised farmers were found. This suggests that the advice may have a greater effect on items that the farmers already had a tendency to agree with.

4.6. Motivations differences between labelled and non- labelled farmers

The results revealed ($p < 0.1$) differences in responses to ten items between the labelled/ non-labelled groups, seven of which were significant at the 0.05 level. The feeling of an '*environmental responsibility (M1)*' and the establishment of ECA due to being '*personally convinced (M6)*' were answered differently by the labelled and non-labelled farmers, with stronger agreement by the labelled farmers. Furthermore, labelled farmers were found to have stronger agreement that ECA '*fits to their philosophy (M7)*' and that it is important to have '*Confidence in effectiveness (M11)*' of the biodiversity promotion. Again, this appears to reinforce the core philosophy behind organic and IP-Suisse production and reflects a mindset that labelled farmers are more motivated to promote biodiversity. This mindset may contribute to an overall greater motivation for labelled farmers to promote biodiversity on their farms, which goes some way to explaining Bengtsson, Ahnström, and

Weibull's (2005) findings, which were confirmed by Tuck et al. (2014), that organic farms host an average of 30% more species than conventional farms.

The finding that advice may have a greater effect on 'farmer conservation motivations scale' items that the farmers already had a tendency to agree with, was not found in the comparison of labelled and non-labelled farmers, in which the group differences seem to be independent of the general agreement with the items.

5. Conclusions

Several of the items from the farmer beliefs scale questions regarding the tasks of farmers received strong agreement from all farmers: regardless of production type or whether they had received targeted advice. These items included: maintaining the *'soil fertility (B1)'* for the next generation; production of *'healthy food (B2)'*; farming in a way that is *'animal friendly (B3)'*; using natural *'resources sustainably (B4)'*; and ensuring *'food security (B5)'* for the Swiss population. Given the high level of agreement, it would have been difficult for advisors to cause a positive effect and it may have been a case of preaching to the converted. An alternative explanation for the absence of differences in responses to these items is that these responses might be a result of socially desirable responding across the entire sample. However, any response bias is likely to have been applied to all questions, and other items, such as *provision of 'Recreation areas (B12)'* and '*appreciation from outside (B14)*', which can also be considered to be socially desirable, received less agreement. A reasonable conclusion is that the belief items that received almost universal agreement are part of the identity of the responding farmers, and that the degree of agreement is a reliable result. These results also cannot be solely attributed to oversampling of organic farmers because the responses from the non-labelled farmers also indicated the strongest agreement with these items, so their ranking in importance would have been high: even without the organic component of the labelled farmers.

Most of the between-group differences between advised and non-advised farmers were found in responses to the 'farmer beliefs scale' items with the least general agreement (mean responses below 3.75). A possible explanation is that advice can have little effect if there is already almost universal agreement with the particular belief. For example, the item: *an important task is maintaining the soil fertility for the next generation (B1),* received a mean response of 4.91, so farmer advice might not have sufficiently improved that already high agreement score for the (smaller) 'advised' group (mean = 4.96) for the difference to be significant. On the other hand, it may be a confirmation that beliefs are resilient and difficult to change (Bandura 1989); especially when the beliefs are strongly held. In contrast, the differences between advised and non-advised farmers in responses to the 'farmer conservation motivations scale' were more pronounced for the items with more universal agreement. The items in this scale had more (than the 'farmer beliefs scale') potential for improvement. For example, the item with the highest mean agreement (mean = 4.18) was the *Environmental responsibility (M1)* item, which increased to mean = 4.64 for the advised farmers. Furthermore, these observations might also indicate that advice has a greater effect on items that the farmers already had a tendency to agree with. It will be the challenge for future research to investigate these relationships more fully.

The aim of this study was to investigate whether beliefs and motivations can be influenced, such as by giving on-farm advice to farmers. The results suggest that the advice received by farmers in this study may have been a sufficient intervention for them to adjust their beliefs (Dweck 2008). However, an alternative explanation is that the difference is due to response bias, with the farmers who agreed to participate in the 'Scoring with biodiversity' project and to accept on-farm advice already being more motivated to implement ECAs. To draw definitive conclusions about the effect of on farm advice, it would have been necessary to measure beliefs prior to the farmers receiving advice, and the absence of such data is a limitation of this study. A reasonable supposition is that the difference is likely to be a mixture of bias and effect although the finding that there were no significant differences between the randomly drawn sample of farmers and the sample of farmers who had participated in the 'Scoring with biodiversity' project adds weight to the conclusion that differences may be an effect of the advice.

More pronounced differences were found between the beliefs held by labelled farmers and those held by their non-labelled producing colleagues. Our explanation for this discrepancy is that the beliefs held by the labelled farmers, which were more positive towards nature protection, were at least part of the motivation for them to convert to organic production or adopt the IP-Suisse label. In other words, the choice of whether or not to adopt the IP-Suisse label or to convert to organic might be dependent on the compatibility with the farmer's individual belief system. However, further research would be required to fully explore this relationship.

A limitation of the study was that the pool of farmers who had received biodiversity advice was only 25: of whom 23 responded to the survey, which limited the total sample size for the study. Furthermore, the proportions of the respondents from the three main production systems: organic, IP-Suisse, and PEP, in the sample do not accurately reflect the proportions in Switzerland. Therefore caution should be taken when making generalizations about Swiss farmers based on the descriptive statistics. Despite these limitations, the sample size was adequate for statistical analysis (Norman 2010), and the between-group comparison can be considered to be resistant to socially desirable response bias because it is reasonable to assume that any bias was constant across responses.

However, despite these limitations, the findings that do not contradict the proposition that targeted advice can indeed influence the beliefs of farmers and group differences between advised and non-advised farmers were found. The findings of stronger beliefs and motivations for implementing ECAs held by farmers who had received biodiversity advice would have been predicted by Chevillat et al. (2017) and Smallshire, Robertson, and Thompson (2004) who each found evidence for the effectiveness of farmer advice, particularly when the advice is targeted rather than neutral (Dweck 2008).

This study has shown that several beliefs and motivations received more agreement by farmers who had been given on-farm advice, which provides some options for farm advisory services. The finding that there were significant differences between the advised farmers and the remainder of the sample for some variables, but not for others, means that farm advisors should choose to either focus their advice on the variables in which the advised group agreed more strongly; or to increase efforts in advice on variables that correlate with the proportion of implemented ECA, but did not receive stronger agreement by the advised farmers.

Disclosure statement

No potential conflict of interest was reported by the authors.

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Review

The challenges of including impacts on biodiversity in agricultural life cycle assessments

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ABSTRACT

Agriculture is considered to be one of the main drivers for worldwide biodiversity loss but the impacts of agricultural production on biodiversity have not been extensively considered in Life Cycle Assessments (LCAs). Recent realisation that biodiversity impact should be included in comprehensive LCAs has led to attempts to develop and implement methods for biodiversity impact assessment. In this review, twenty-two different biodiversity impact assessment methods have been analysed to identify their strengths and weaknesses in terms of their comprehensiveness in the evaluation of agricultural products. Different criteria, which had to meet the specific requirements of biodiversity research, life cycle assessment methodology, and the evaluation of agricultural products, were selected to investigate the identified methods. Very few of the methods were developed with the specific intention of being used for agricultural LCAs. Furthermore, none of the methods can be applied globally while at the same time being able to differentiate between various agricultural intensities. Global value chains and the increasing awareness of different biodiversity impacts of agricultural production systems demand the development of evaluation methods that are able to overcome these shortcomings. Despite the progress that has already been achieved, there are still unresolved difficulties which need further research and improvement.

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Contents

1. Introduction

Life Cycle Assessments (LCAs) investigate the entire life cycle of a product or production system, including raw material acquisition,

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production, use, and finally, recycling and disposal (International Standard Organisation, 2006). They are increasingly used for estimating the environmental efficiency of different product systems and for calculating and comparing the environmental impacts of agricultural production (Roy et al., 2009; Müller-Lindenlauf et al., 2010; Yan et al., 2011; Milà i Canals et al., 2012, O'Brien et al., 2012; Tuomisto et al., 2012). LCAs quantify the environmental impacts of different factors; with energy consumption, climate, ozone formation, eutrophication, acidification, land use and eco-and human toxicity included as impact categories in the majority of agricultural LCAs (Meier et al., 2015). However, despite early awareness that land use related impacts on biodiversity should be included in a thorough environmental impact assessment of agricultural processes and products (Geier, 2000), impacts on biodiversity are often omitted (Hayashi et al., 2006).

Efforts to address this lack have recently led to the development and improvement of biodiversity impact assessment methods that provide specific information about the change of biodiversity in particular land use processes. However, authors have continually pointed out that biodiversity assessment in LCAs in general (Milà i Canals et al., 2007, Curran et al., 2010; de Baan et al., 2013b; Koellner et al., 2013b; Souza et al., 2014; Teixeira et al., 2015), and in agricultural contexts in particular (Urban et al., 2012), have not yet been sufficiently developed. Curran et al. (2010) noted several conceptual shortcomings in biodiversity assessment in LCAs; including that the scale is often not considered, functional and structural aspects are underrepresented, and taxonomic and geographic coverage is usually insufficient. Souza et al. (2014), in their review of existing biodiversity impact assessment methods, revealed that landscape effects are often neglected and that the complexity of natural dynamics and processes are inadequately reflected. Souza et al. (2014) discussed many unresolved issues with respect to land use-related impacts on biodiversity in LCAs, but they did not consider the particularities of agricultural land use. However, agriculture is one of the main drivers of the worldwide biodiversity loss (Hole et al., 2005; Benton, 2007; Feber et al., 2007; Kleijn et al., 2009) so impacts on biodiversity are particularly relevant for LCAs of agricultural products.

When developing a particular biodiversity impact assessment method for LCAs in general, the chosen methodology should be consistent with the state of research on biodiversity assessment. For a thorough biodiversity assessment, it is necessary to cover several aspects of biodiversity and choose the appropriate indicators for each of these aspects. The principle difficulties of biodiversity assessment include that biodiversity cannot be measured directly and that the very different aspects of biodiversity mean that measurement requires a set of indicators rather than a single indicator (Bolger, 2001; Büchs, 2003; Stoddard et al., 2006; Nielsen et al., 2007; Flynn et al., 2009; van Strien et al., 2009; Heink and Kowarik, 2010; Beck et al., 2012; Mandelik et al., 2012). Specific difficulties of indicator-based approaches include: choosing the appropriate biodiversity metric so that indicator/environment relationships are transferable (Beck et al., 2012); and selecting the appropriate scale (field, farm, landscape) (Feld et al., 2009; Batáry et al., 2012; McMahon et al., 2012), which is important because the responses of different taxa can vary depending on the chosen scale (Gabriel et al., 2010).

LCAs in the agricultural context have some additional specific considerations. It is necessary to distinguish between planned biodiversity, which is under the direct management influences of the farmer (e.g. crops, livestock, etc.), and associated biodiversity, which is the immigrating and colonizing biodiversity from the surroundings (Altieri, 1999; Swift et al., 2004; Tscharntke et al., 2012). Global applicability is particularly important with the assessment of agricultural products because they commonly have a global life cycle. Furthermore, the choice of an appropriate functional unit for the assessment of agricultural products, (above all, area per time or mass of product) is critical (Geier, 2000). The difference in the impact on biodiversity between organic and conventional agriculture has been frequently studied (Hole et al., 2005; Batáry et al., 2013; Garibaldi et al., 2013; Schneider et al., 2014; Tuck et al., 2014) and Billeter et al. (2008), Flynn et al. (2009), Kleijn et al. (2009) and Hawes et al. (2010) each showed the varying impacts on biodiversity of different agricultural systems or levels of intensity. Therefore, particularly in comparative LCAs of agricultural products, impact assessment on biodiversity needs to be sufficiently sensitive to different agricultural intensities.

The aim of this study is to review existing impact assessment methods for biodiversity that are used in LCAs, and to evaluate their suitability for application in LCAs in agricultural contexts. We provide an overview of the methodological developments and improvements and identify their strengths and weaknesses. We focus on the selection of biodiversity aspects, indicators, reference conditions, functional units and the possibility of differentiating between several agricultural intensities. The paper concludes with recommendations on how to overcome identified shortcomings.

2. Methods

We used the search terms "biodiversity" and "life cycle assessment" in the electronic databases of "ISI Web of Science" to identify literature that was relevant to biodiversity impact assessment methods in LCAs. The literature search was extended by including studies that were cited in the identified papers. The biodiversity impact assessment methods reported in the literature were analysed and compared with respect to their suitability for the assessment of agricultural products.

For the analysis, we evaluate the suitability of applying biodiversity impact assessment methods in LCA, based on the following criteria:

- Biodiversity aspects, which refer to specific partitions of biodiversity that are the focus of different assessment methods because biodiversity is not measurable as a whole.
- Biodiversity indicators, which are measures or metrics based on verifiable data that convey information to express size, amount or degree of biodiversity (Biodiversity Indicators Partnership, 2011)
- Reference conditions and functional units, which are chosen in the practical application of biodiversity impact assessment methods in LCAs. We especially focus on the extent to which they are suitable for assessing the biodiversity impacts of agricultural products.
- Geographic applicability, which includes the range of contexts, up to global, in which the methods can be applied
- The ability of the methods to distinguish between different agricultural intensities.

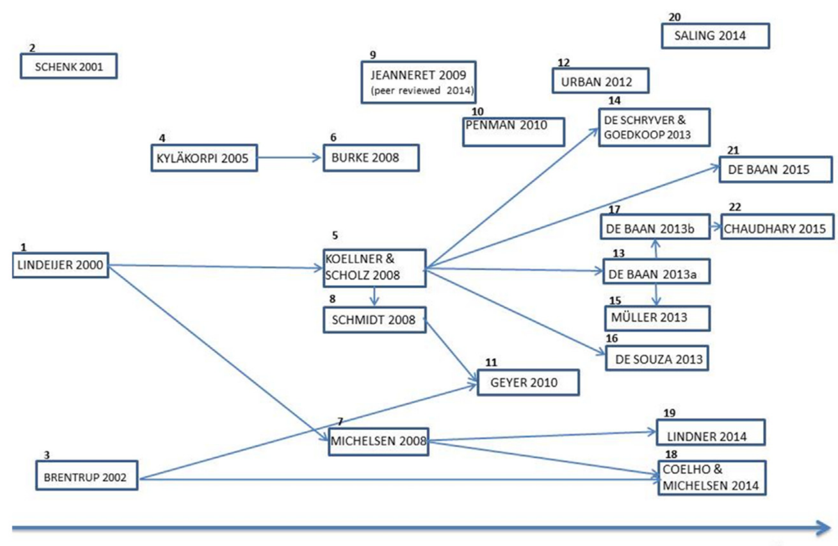
3. Results and discussion

3.1. Origin of papers

We identified seventeen studies that were published in peer-reviewed journals between 2000 and 2015. Furthermore, we identified three relevant reports (Kyläkorpi et al., 2005; Jeanneret et al., 2009; De Schryver and Goedkoop, 2013) and two recent conference papers (Lindner et al., 2014; Saling et al., 2014). Several of the methods have been further used or elaborated in following studies. The spectrum of the studies ranged from concepts and approaches to applicable methods or even to the provision of characterization factors (CFs). An overview of the development of biodiversity impact assessment methods in LCA is shown in Fig. 1.

The seminal publication appears to be that of Lindeijer (2000), which differentiated between “transformation impacts” and “occupation impacts”, and which has been adopted in thirteen subsequent studies. During the time period between 2000 and 2008, the methods of Koellner and Scholz (2008) and Michelsen (2008) were built directly on Lindeijer (2000). The methods or concepts of Schenck (2001), with land use and biodiversity indicators compiled at an expert workshop; Brentrup et al. (2002), which is based on the concept of hemeroby; and Kyläkorpi et al. (2005) were independently developed at the same time. The report by Kyläkorpi et al. (2005) describes the “Biotope Method“ in which biotope areas are assessed before and after land use changes to evaluate the impacts of the change, which was later applied by Burke et al. (2008). The method of Brentrup et al. (2002) and the work of Schmidt (2008) provided the basis for the study by Geyer et al. (2010a). Koellner and Scholz (2008) in turn was the basis for the methods of Schmidt (2008), de Baan et al. (2013a), de Souza et al. (2013) and de Baan et al. (2015). De Schryver and Goedkoop (2013) also took over some aspects from Koellner and Scholz (2008). The methods of de Baan et al. (2013b) and Mueller et al. (2013) built on de Baan et al. (2013a). Chaudhary et al. (2015) proposed an advancement of the work of de Baan et al. (2013b). The methods of Lindner et al. (2014) and Coelho and Michelsen (2014) both rely on Michelsen (2008) with Coelho and Michelsen (2014) also adopting aspects of the method developed by Brentrup et al. (2002). In the time period 2009–2014, the more independent ideas or concepts of Jeanneret et al. (2009), Penman et al. (2010), Urban et al. (2012) and Saling et al. (2014) were developed. Jeanneret et al. (2009) and Saling et al. (2014), used biodiversity as an independent impact category; Penman et al. (2010) used a single biodiversity measure that was generated through a scoring system and incorporated in LCAs.; and Urban et al. (2012) combined the spatially specific procedure of environmental planning with the product-specific approach of LCAs. Jeanneret et al.’s (2009) report describes SALCA-Biodiversity, which is a biodiversity assessment method for agricultural LCAs that has been used for LCAs in different contexts (Köpke and Nemecek, 2010; Nemecek et al., 2011a, 2011b, 2012).

Land use is the foundation and land use changes are the elementary flows commonly considered for biodiversity impact assessment in LCA. The UNEP SETAC Life Cycle Initiative (Koellner



Legend

	Author	Title	Objective of the study
1	**Lindeijer (2000)**	Biodiversity and life support impacts of land use in LCA	Development of an applicable method for assessing land use impacts on ecosystems in LCAs.
2	**Schenck (2001)**	Land use and biodiversity indicators for life cycle impact assessment	Providing a set of indicators, which describe the impacts of land use worldwide at all scales for the use of LCAs.
3	**Brentrup et al. (2002)**	Life cycle impact assessment of land use based on the hemeroby concept	The objective of the paper is the assessment of environmental impacts of land use in LCA studies.
4	**Kyläkorpi et al. (2005)**	THE BIOTOPE METHOD 2005: A method to assess the impact of land use on biodiversity	Presenting a method which was developed to quantify the biodiversity impacts trough land use for generating electricity or heat products, in a case study of Vattenfall group in Sweden.
5	**Koellner & Scholz (2007, 2008)**	Assessment of land use impacts on the natural environment. Part (1 and 2)	Development of a model with damage functions and generic characterisation factors to quantify damages on ecosystems through land occupation and land transformation.
6	**Burke et al. (2008)**	Testing an Scandinavian biodiversity assessment tool in an African desert environment	Testing of the “Biotop Method” (Kyläkorpi et al. 2005), a Scandinavian biodiversity assessment tool, in an African desert environment.
7	**Michelsen (2008)**	Assessment of land use impact on biodiversity	Providing a methodology to include the land use impacts on biodiversity in LCAs with a case study of forest operations in Norway.

Fig. 1. Overview of the biodiversity impact assessment method development. Arrows indicate studies that build on the preceding studies.

8	**Schmidt (2008)**	Development of LCIA characterisation factors for land use impacts on biodiversity	Development of a method to determine characterization factors for any land use type and any region without the need of overwhelming data requirements.
9	**Jeanneret et al. (2009, 2014)**	Method for the assessment of the impacts of agricultural activities on biodiversity in LCAs (SALCA Biodiversity)	Development of a method for including the impact category biodiversity in LCAs of agricultural processes.
10	**Penman et al. (2010)**	A proposal for accounting for biodiversity in life cycle assessment	An alternative approach based on expert opinions for incorporating biodiversity impacts into LCAs
11	**Geyer et al. (2010a, 2010b)**	Coupling GIS and LCA for biodiversity assessment of land use (Part 1 and 2)	An approach for coupling GIS and LCA for the biodiversity assessment of land use applied in a case study of ethanol production in California.
12	**Urban et al. (2012)**	Spatially differentiated examination of biodiversity in LCA on national scale exemplified by biofuels.	Development of a new impact assessment method for biodiversity, where spatial references can be included.
13	**De Baan et al. (2013a)**	Land use impacts on biodiversity in LCA: a global approach	Quantification of land use impacts (biodiversity damage potential) across different world regions.
14	**De Schryver & Goedkoop (2013)**	ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level (Impacts of land use)	Implementation of an LCIA method that is harmonised in terms of modelling principles and choices, but which offers results at both the midpoint and the endpoint level.
15	**Mueller et al. (2013)**	Comparing direct land use impacts on biodiversity of conventional and organic milk based on a Swedish case study	Demonstration of a method to quantify and compare direct land use impacts on biodiversity of organic and conventional food products (milk).
16	**De Souza et al. (2013)**	Land use impacts on biodiversity in LCA: proposal of characterization factors based on functional diversity	A method is proposed where a functional diversity index is used to calculate characterization factors for land use impacts.
17	**De Baan et al. (2013b)**	Land use in life cycle assessment: global characterization factors based on the regional and global potential species extinction	An approach to model the impacts of regional land use on plants, mammals, birds, amphibians and reptiles. Calculation of the total potential damage caused by all land uses within an ecoregion and the allocation of the total damage to different land use types per ecoregion.
18	**Coelho & Michelsen (2014)**	Land use impacts on biodiversity from kiwifruit production in New Zealand assessed with global and national datasets	A methodology for assessing the impact of land use on biodiversity using ecological structures as opposed to information on number of species.
19	**Lindner et al. (2014)**	Proposal of a unified biodiversity impact assessment method	A method, which allows a broad-brush and a detailed assessment of critical, processes and enables LCA users to generate product-related biodiversity impact information.
20	**Saling et al. (2014)**	Assessment of biodiversity within the holistic sustainability evaluation method of AgBalance	Ag Balance™ is a new LCA System to assess the sustainability performance of agricultural goods including the impact assessment of agricultural activity on biodiversity.
21	**De Baan et al. (2015)**	High –resolution assessment of land use impacts on biodiversity in life cycle assessment using species habitat suitability models	Development of a high-resolution assessment method for land use impacts on biodiversity based on habitat suitability models of mammal species.
22	**Chaudhary et al. (2015)**	Quantifying land use impacts on biodiversity: Combining species- area models und vulnerability indicators	Providing an updated impact assessment approach and characterization factors for regional and global biodiversity loss.

Fig. 1. (*continued*).

et al., 2013a, 2013b) provides principals and guidelines with respect to the assessment of land use and influenced many of the newer method developments (de Baan et al., 2013a; de Baan et al., 2013b; de Souza et al., 2013; Mueller et al., 2013). Lindeijer's (2000) differentiation between "transformation impacts" and "occupation impacts" thereby also provides the roots for the "UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA" (Koellner et al., 2013b). The UNEP SETAC Life Cycle Initiative also started a flagship project with the aim of developing global guidelines *inter alia* for land use impacts on biodiversity (Milà i Canals et al., 2014). Within this project, Souza et al. (2014) analysed the commonly applied methods for assessing biodiversity loss caused by land use. They identified limitations of the existing methods, including difficulties in applying biodiversity indicators, the importance of defining the reference condition, and limitations of applied ecological models (Souza et al., 2014). Recommendations for addressing these difficulties were formulated in expert workshops and summarised by Teixeira et al. (2015), who concluded that indicators for local land use impacts should be representative and quantitative; and the regional level should be considered so that intrinsic biodiversity values and threats to biodiversity can be expressed relative to other areas. In a recent FAO report, indicators and life cycle assessment methods were reviewed in terms of their application to livestock production systems (LEAP, 2015).

3.2. Functional unit

All of the identified studies focussed on land use impacts on biodiversity, so it was not surprising that eleven studies used land area as functional unit (Lindeijer, 2000; Brentrup et al., 2002; Koellner and Scholz, 2008; Schmidt, 2008; Jeanneret et al., 2009; Geyer et al., 2010a; Urban et al., 2012; de Baan et al., 2013a; de Baan et al., 2013b; De Schryver and Goedkoop, 2013; de Souza et al., 2013). Brentrup et al. (2002) also used area as the functional unit but commented that a product related functional unit might be better suited to agricultural LCAs because it would reveal any efficiency differences between land use types. Several studies instead used product-related functional units, with Michelsen (2008) using the volume of logs, Mueller et al. (2013) using weight of energy corrected milk; and Kyläkorpi et al. (2005) using energy expressed in kWh. Burke et al. (2008) applied the method developed by Kyläkorpi et al. (2005) and noted that other functional units are equally useful, such as weight of mined uranium or area. The authors concluded that the method is thus particularly suitable for LCAs but did not further specify how the other functional units could be operated. Product weight is the base unit in the methods developed by Lindner et al. (2014), de Baan et al. (2015) and Chaudhary et al. (2015) and was also used in case studies for kiwi production (Coelho and Michelsen, 2014) and oilseed rape production (Saling et al., 2014). Penman et al. (2010) offered no information about the functional unit chosen in their study.

The dominant factor in the selection of the functional unit is the question behind the biodiversity assessment. The results for the biodiversity impacts for intensive or extensive agricultural production systems may differ depending on the functional unit chosen. For example, an extensive production system may show a lower biodiversity impact per area but a higher impact per production unit, than a more intensive production system. A product related functional unit provides valuable information of the production efficiency of a system, but it can be questioned whether this information is sufficient in terms of the biodiversity assessment of agricultural production. In systems with a high production output for instance, the biodiversity damage, which is related to the area, could be masked when the impact related to the product output is relatively low. Biodiversity assessment must be site-specific, as biodiversity is always related to the site, so a specific area can serve as functional unit.

For the assessment of agricultural systems some peculiarities relating to product efficiency should be considered. In agricultural production, the highest possible yield shouldn't be the only goal that is traced but, as is demanded from agricultural policy in most countries, the multi-functionality of agriculture should also be considered (OECD, 2001; EU Comission, 2003; OECD, 2003). Milà i Canals (2003 p.29) pointed out that "agriculture systems are multi-functional and so it is difficult to define which functional unit is delivered by the system." Therefore an exclusive use of a product-related functional unit will rarely be sufficient from the perspective of multifunctionality in agriculture.

3.3. Biodiversity aspects and indicators

In twelve studies, the biodiversity impact assessment methods cover the biodiversity aspect "species diversity" of specific indicator species (see Table 1). Seven studies looked at the biodiversity aspect "biotope quality". The aspect "functional diversity" was addressed in two cases (Lindeijer, 2000; de Souza et al., 2013). The aspect "endangered or threatened species" was also included in some of the studies (Schenck, 2001; Koellner and Scholz, 2007, 2008; Jeanneret et al., 2009, 2014; Penman et al., 2010; Jeanneret et al., 2014; Chaudhary et al., 2015; de Baan et al., 2015). "Genetic diversity" was not considered in any of the analysed methods, although this aspect could play an important role; especially in agriculture in terms of crop varieties and diversity of breeds. Lindner et al. (2014) and Saling et al. (2014) stated the intention of assessing "total biodiversity" within their methods. In the case of Lindner et al. (2014) this relies on the evaluation of experts. Surrogates or indicators for total biodiversity do not exist so it remains unclear which aspects will be considered in the expert assessment.

Most studies do not explain why a particular biodiversity aspect, such as species diversity, was chosen (Michelsen, 2008; Jeanneret et al., 2009; de Baan et al., 2013a). However, such an explanation would increase the transparency of the biodiversity assessment. An exception is de Souza et al. (2013) who point out the advantages of using functional diversity compared to species richness in their paper; for example its suitability as an indicator for the biodiversity aspect "biodiversity loss". The selection of which indicator to use has received more attention.

The reasons for the selection of already-used indicators include data availability (Koellner and Scholz, 2008; Geyer et al., 2010a; Urban et al., 2012; de Baan et al., 2013a; de Baan et al., 2013b; de Souza et al., 2013), use in previous methods (Schmidt, 2008; Geyer et al., 2010a; de Baan et al., 2013a), acceptance by LCA practitioners (de Baan et al., 2013a), and knowledge of the taxa and the suitability for the assessment of agricultural impacts (Jeanneret et al., 2009). However, the correlation between the biodiversity aspect and the selected indicators was not always apparent or clearly defined. Heink and Kowarik (2010) recommend that the choice of indicator should comply with the suitability for the indicated "biodiversity aspect", but this reason was not mentioned in the reviewed studies.

Species richness of vascular plants is frequently used as an indicator, which is based on the assertion by Duelli (1997) and Duelli and Obrist (1998) that it correlates with the species richness of other species groups. However Souza et al. (2014) suggest that this assumption should be made with caution: a position that would be supported by (Billeter et al., 2008; Rossi, 2011; Jeanneret et al., 2012). de Baan et al. (2013a) lists four disadvantages of using species richness as an indicator; including dependency on the sample efforts, missing information on species abundance, neglect of the species conservation status, and the lack of consideration of conservation targets. In de Souza et al. (2013) the basis for the assessment of functional biodiversity is the "Petchey and Gaston's FD index" (Petchey and Gaston, 2002) where a set of functional traits is combined. Jeanneret et al. (2009) and Penman et al. (2010), in contrast, used a system of biodiversity points or scores that were calculated on the basis of published data and literature, in combination with expert opinions.

3.4. Reference condition

Assessment of impacts on biodiversity implies an assessment of change in biodiversity as a result of a particular action, which in turn implies that a reference condition is needed. Furthermore, reference conditions in LCAs meet two functions, which are rarely distinguished in the literature. One function is to serve as a benchmark to quantify the biodiversity impact by comparing it with the chosen reference state (point in time, desired or targeted status or average land use). The second function can be regarded as a normalization step, which is needed to evaluate the biodiversity impact across different biogeographic regions. This means that a common reference is established for biodiversity evaluation, which is used in all regions to be evaluated. Reference conditions in the identified literature could be classified into five different groups, which are shown in Table 2:

Koellner et al. (2013b) recommended that reference conditions should be based on either the theoretical construct of potential natural vegetation (PNV) (Chiarucci et al., 2010); a quasi-natural state of the considered region (biome, ecoregion, etc.); or the current land use mixture which shows both the positive and negative impacts. However, the disadvantage of PNV as reference, which is used by Brentrup et al. (2002), de Baan et al. (2013a), de Baan et al. (2013b), de Souza et al. (2013), is that it is fictional. This means that there is the need to define another real state, which best approximates this fictional condition (e.g. (Semi-) natural vegetation) to get data for the biodiversity assessment. Augustin et al. (2007), conducted an expert workshop and concluded that PNV is an unsuitable reference for the assessment of agricultural ecosystems. Souza et al. (2014) described the difficulty of comparing natural and PNV states and noted that they fail to consider possible active land restoration, which is a further disadvantages of the use of PNV as

Table 1
Biodiversity aspects, indicators and measures used in the analysed studies.

Study	Type of biodiversity aspects	Indicator	Biodiversity measure
Lindeijer (2000)	Species diversity, Life support function (functional diversity)	Species diversity of vascular plants, free net primary biomass production (fNPP)	Species number, Amount of biomass per area and year
Schenck (2001)	Habitat quality/diversity, Species diversity, Soil quality, Rare or threatened species, Functional diversity, Invasive species, Fragmentation, Native Vegetation, Habitat distribution/connectivity	Protection of priority habitat/species Soil organic matter Proximity to & protection of high priority vegetative communities Interface between water and terrestrial habitats/buffer zones Assimilative capacity of water and land. Hydrological function Coverage of invasive species within protected areas Road density Percent native- dominated vegetation Restoration of native vegetation Adaption of Best Management Practise linked to biodiversity objectives Distribution (patchiness, evenness etc.) Connectivity of native habitat	Area of habitat that is physically protected; habitats 100 feet each side of rivers, habitats of threatened and endangered species. Concentration of carbon in the soil. Area of habitat set aside that is identified as "high priority" in The Nature Conservancy vegetative maps Total linear space of aquatic habitat protected via physical means vs. total area managed. Depletion of water resources For physically protected areas, density of non-native vegetation Units of road per square units Area in native species dominated areas/total area managed Area newly returned to native habitat Number of Best Management Practices adopted Size of native- managed acres vs. total area managed, size of native-managed acres vs. average field size On managed areas, percent of native- managed land units that have at least one adjacency to other native managed land.
Brentrup et al. (2002)	Biotope quality, Naturalness	Hemeroby* *an indicator for the human influence on ecosystems (Kowarik, 1999)	Naturalness Degradation Potential (NDP)
Kyläkorpi et al. (2005)	Biotope quality, Biotope diversity	Gains and losses of biotope categories	Area change of different biotope categories
Koellner and Scholz (2007, 2008)	Species diversity, Threatened species	Species richness of vascular plants, Mosses, Molluscs	Species number
Burke et al. (2008)	Biotope quality Biotope diversity	Gains and losses of biotope categories	Area changes of different biotope categories
Michelsen (2008)	Biotope quality, Loss of biodiversity	Ecosystem Scarcity (ES) Ecosystem Vulnerability (EV) Conditions for Maintained Biodiversity (CMB)	ES after (Weidema and Lindeijer, 2001) EV after (Weidema and Lindeijer, 2001) or (Peter et al., 1998) CMB after (Larsson, 2001)
Schmidt (2008)	Species diversity	Species richness of vascular plants	Species number
Jeanneret et al. (2009, 2014)	Total species diversity, diversity of stenoecious species threatened/rare species	Flowering plants, birds, small mammals, amphibians, snails, spiders, carabid beetles, butterflies, grasshoppers, bees and bumblebees	Biodiversity points (ordinal scale 1–5) derived from expert opinion and literature.
Penman et al. (2010)	Threatened and endemic species, endangered ecological communities	Different taxa (not further specified)	Biodiversity scores (ordinal scale -10-10) derived from published data and expert opinion
Geyer et al. (2010a, 2010b)	Biotope quality, Species diversity	Hemeroby, reptiles, mammals, birds, amphibians	Naturalness Degradation Potential (NDP), Species richness, species abundance, evenness
Urban et al. (2012)	Biotope quality	Claiming of land that is particularly valuable with respect to biodiversity (share/ha) Cultivation proportion of agricultural crops Nitrogen fertilizer (kg N/ha).	Valuable agricultural land Modified Shannon Weaver Index (Cultivation proportion of agricultural crops). Impact Grades
de Baan et al. (2013a)	Loss of Species diversity	Species richness	Species number
De Schryver and Goedkoop (2013)	Loss of Species diversity	Species richness of plants	Species number
Mueller et al. (2013)	Loss of Species diversity	Species richness of vascular plants	Species number
de Souza et al. (2013)	Loss of functional diversity	Mammals, birds, plants	Petchey and Gaston's FD Index
de Baan et al. (2013b)	Loss of Species diversity	Species richness and endemism of mammals, amphibians, reptiles, birds, species richness of plants	Species number
Coelho and Michelsen (2014)	Biotope quality	Ecosystem Scarcity (ES) Ecosystem Vulnerability (EV) Conditions for Maintained Biodiversity (CMB)/ Hemeroby	ES after (Michelsen, 2008) EV after (Weidema and Lindeijer, 2001), (Peter et al., 1998) or (Michelsen, 2008) Naturalness Degradation Potential (NDP) Numerical values derived from expert opinions

Table 1 (*continued*)

Study	Type of biodiversity aspects	Indicator	Biodiversity measure
Lindner et al. (2014)	Total Biodiversity depending on national biodiversity strategies and expert opinions	Choice of indicator depending on national biodiversity strategies and expert opinions	
Saling et al. (2014)	Total Biodiversity	**Pressure Indicators:** Low cropping diversity Nitrogen surplus Ecotoxicity potential of pesticides High farming intensity Outcrossing potential **State Indicator:** IUCN Assessment (Farmland Bird populations) **Response Indicators:** High cropping diversity Agri-environment schemes Protected areas Low farming intensity	Calculation factors (Depending on indicator specific algorithms)
de Baan et al. (2015)	Loss of Species diversity, threatened and rare species	Species richness, threat level and rarity of mammals	Species number
Chaudhary et al. (2015)	Loss of Species diversity, threatened and endemic species	Species richness and threatened endemic richness of mammals, amphibians, reptiles, birds, species richness of plants	Species number

Table 2
Reference conditions.

Type of reference condition	Number of studies	Authors
Alternative use scenario	3	(Jeanneret et al., 2009; Urban et al., 2012; Saling et al., 2014)
Average land use mix	4	(Lindeijer, 2000; Koellner and Scholz, 2008; Geyer et al., 2010a; de Baan et al., 2015)
Natural or semi -natural reference	10	(Brentrup et al., 2002; Michelsen, 2008; Schmidt, 2008; de Baan et al., 2013a; de Baan et al., 2013b; De Schryver and Goedkoop, 2013; de Souza et al., 2013; Mueller et al., 2013; Chaudhary et al., 2015; de Baan et al., 2015)
Temporal or historical reference	3	(Kyläkorpi et al., 2005; Burke et al., 2008; Penman et al., 2010)
Desired state/maximum biodiversity potential	2	(Coelho and Michelsen, 2014; Lindner et al., 2014).

reference. Teixeira et al. (2015) point out that there is no agreement regarding a reference situation. In the FAO report, the authors summarize that it is not possible to account for beneficial biodiversity impacts with the choice of reference conditions in most LCA studies (LEAP, 2015).

The main determinate for the selected reference is the context for the biodiversity assessment. Selection of an historical reference situation, as in Penman et al. (2010) is problematic because of the poor data availability for historical points in time. One of the two reference states used by Lindeijer (2000) for the indicator species diversity of vascular plants was the number of species found in the least disturbed situation in a region, while Koellner and Scholz (2008) used the average species richness in a region. The disadvantage of this approach is that the reference state only covers the biodiversity aspect; “species richness”.

No reference state is indicated in SALCA Biodiversity (Jeanneret et al., 2009), so the method is only suitable to show changes in comparison with an alternative use scenario, while Urban et al. (2012) and Saling et al. (2014) used alternative use scenarios as their reference conditions. Natural or semi- natural conditions were used as reference conditions in several studies (Brentrup et al., 2002; Michelsen, 2008; Schmidt, 2008; de Baan et al., 2013a; de Baan et al., 2013b; De Schryver and Goedkoop, 2013; de Souza et al., 2013; Mueller et al., 2013; Chaudhary et al., 2015; de Baan et al., 2015). However, there is considerable variation in terms of biodiversity between different natural or semi- natural habitats so their suitability as a reference condition for the biodiversity assessment of agricultural ecosystems is open to discussion. For example, a habitat that is low in biodiversity but contains rare endemic species may be difficult to compare with a habitat that is high in biodiversity; although both may be natural or semi natural habitats. This approach, however, corresponds to the LCA thinking; with the comparison of a non- use of the biodiversity resource. In two studies a desired state or potential of biodiversity is used as the reference condition (Coelho and Michelsen, 2014; Lindner et al., 2014). de Baan et al. (2015) assert that, with their method, different reference situations can be used for the first time. In a case study they compared the reference situations: ‘current land use’ and ‘maximum species range (natural land)’.

Independently the question behind the biodiversity assessment should determine the choice of the reference condition. There is no reference condition which is suitable for all questions concerning the biodiversity assessment of agricultural products. Hence we recommend a comparison of different scenarios in a second step, where for example the actual land use mixture or an alternative use scenario etc. can be used. Reference condition is inherently local, and the concept is relatively straightforward when considering a particular land use, but a thorough LCA should also consider inputs and effects that are not necessarily local.

3.5. Global applicability of the methods

It remains difficult to assess the biodiversity impacts of different agricultural production systems; particularly when parts of the production are not located in the same country or even in the same biogeographic region. The globalization of agricultural markets resulted in a growing specialization of agricultural farms. This means that the products consist of production inputs, which are often produced in different parts of the world. The implication of this is that impact assessment methods should be applicable in a wide range of contexts, if not globally. We consider a biodiversity assessment method is globally applicable when comparability of the biodiversity impact assessment is achieved on a global level. This means that it is possible to assess products with global value chains so that the individual compounds can be evaluated under the same conditions. With a globally applicable method, it should be possible to derive global characterization factors (CFs). The CF is a numerical value that describes the impact size and is included in the impact assessment of LCAs.

Koellner et al. (2013a) and de Baan et al. (2013a) each argue that a system of regionalized assessment is needed because of the spatial heterogeneity of biodiversity but that LCAs with global value chains require a global scale; with Koellner et al. (2013a) recommending a standardization of both systems. Characterization factors are relatively easy to generate in such classification systems, which was done in de Baan et al. (2013a), de Baan et al. (2013b) and Chaudhary et al. (2015). In this review, we define criteria; namely the land use or land cover classification, the biogeographic reference and data availability that can be combined to assess the global applicability of the different approaches.

The use of a global **land use and land cover system** simplifies the global applicability because the world is classified with a uniform standard. Brentrup et al. (2002) and Urban et al. (2012) used the Corine Land Cover data (European Environmental Agency, 2000), which differentiates between five major land cover types (artificial surfaces, agricultural areas, forests and semi-natural areas wetlands, and water bodies). Koellner and Scholz (2008), Koellner et al. (2013a) each used an extension of the Corine classification system that includes a distinction between high and low intensities of agriculture and forestry. de Baan et al. (2013a) and de Souza et al. (2013) also refer to an extended classification system derived from Koellner et al. (2013a). de Baan et al. (2015) used the land cover classes (63) from GlobCover v2.3 (European Space Agency, 2010) and Coelho and Michelsen (2014) used a land cover database (LCDB2) from New Zealand.

Several authors created their own classification systems. Schmidt (2008) classified land into arable land (productive land); raw material extraction; urban and sealed land; and nature (nonproductive land) and used FAO data to distinguish between high and low intensity land use. De Schryver and Goedkoop (2013) use the 18 land use types of the ReCiPe method (Goedkoop et al., 2013). de Baan et al. (2013b) differentiate between five broad land use types (agriculture, pasture, managed forests, urban area, and natural habitat), while Chaudhary et al. (2015) distinguished between 6 land use types (intensive forestry, extensive forestry, annual crops, permanent crops, pasture and urban). Jeanneret et al. (2009) used a Swiss specific system with a combination of land cover and use intensity. Lindeijer (2000) used a physiotope classification that is based on important abiotic factors and can be applied globally. In several of the methods, no land cover or land use system was used or proposed (Lindeijer, 2000; Michelsen, 2008; Penman et al., 2010; Lindner et al., 2014; Saling et al., 2014).

Some of the studies do not use a **global available grid for their biogeographic referencing** but instead classify land according to habitat types (Kyläkorpi et al., 2005; Burke et al., 2008; Koellner and Scholz, 2008; Schmidt, 2008; Jeanneret et al., 2009; Penman et al., 2010; Urban et al., 2012; De Schryver and Goedkoop, 2013; Saling et al., 2014). Geyer et al. (2010a) used the regional classification of 59 habitat types in California. In the Biotope method (Kyläkorpi et al., 2005; Burke et al., 2008), four biotope types for a land use cover classification were distinguished; critical, rare and general biotopes, and technotope). de Baan et al. (2013a) and Mueller et al. (2013) used the biomes system, in which the terrestrial world is classified in 14 different biomes and 867 ecoregions, according to (Olson et al., 2001). Olson et al.'s (2001) ecoregions were also used by Michelsen (2008), Geyer et al. (2010a), de Baan et al. (2013b), de Souza et al. (2013), Coelho and Michelsen (2014) Lindner et al. (2014) and Chaudhary et al. (2015). Brentrup et al. (2002) used the classification of eleven biogeographic regions in Europe (European Environmental Agency, 1998).

The lack of **availability of global biodiversity data** is another hurdle for an unlimited application of the methods. It is sometimes possible to provide a relatively detailed assessment of the biodiversity impact from agricultural production on a national level but it is often impossible to compare the biodiversity impacts of the components that are needed for the production; especially when they come from other regions. Methods of biodiversity assessment that are claimed to be globally applicable are often found to be relatively coarse with impacts assessed at the level of different land use types (de Baan et al., 2013b; Chaudhary et al., 2015; de Baan et al., 2015). An overview of an evaluation of the global applicability of each of the methods is given in Table 3. In this table, we have also classified each of the methods according to the effort of the transferability towards a global application. This classification is based on the extent to which the land use or land cover system and the biogeographic reference system are transferable and the data are globally available.

The methods presented by de Baan et al. (2013b), Chaudhary et al. (2015) and de Baan et al. (2015) are globally applicable and perform best in that they are based on global available data (World Wildlife Fund, 2006; Rondinini et al., 2011), and use global applicable land use or land cover systems and a global biogeographic reference system. In principle every method can be applied on a global level with sufficient adjustments and the availability of the required data, but detailed and globally applicable biodiversity assessment of agricultural products has high demands for data availability, which includes detail on various intensities because of differing biodiversity impacts.

It may be sufficient to use a method that is not globally applicable when production processes take place only in one region (products without a global life cycle). In such cases, it may be possible to adapt some parameters for the assessment in other similar regions (Kyläkorpi et al., 2005; Burke et al., 2008) but a normalization step would be needed to achieve the required comparability. In modern agriculture, it is however rare for the entire production, and thus all the biodiversity impacts, to take place in only one region. Data availability is a limiting factor for a global application in the majority of the methods (Brentrup et al., 2002; Kyläkorpi et al., 2005; Burke et al., 2008; Koellner and Scholz, 2008; Schmidt, 2008; Jeanneret et al., 2009; Geyer et al., 2010b; Urban et al., 2012; De Schryver and Goedkoop, 2013; de Souza et al., 2013; Mueller et al., 2013; Saling et al., 2014). de Baan et al. (2013a), Michelsen (2008), and Coelho and Michelsen (2014) resort at least partially to global available data. In the method of Penman et al. (2010) and Lindner et al. (2014) expert interviews are used to generate the necessary information.

3.6. Differentiating between agricultural intensities

A detailed assessment of different agricultural land use activities and intensities is possible with only few of the biodiversity assessment methods that have been developed so far (Jeanneret et al., 2009; Saling et al., 2014). Mueller et al. (2013), in their comparison of organic and conventional milk production, show a difference in the biodiversity impacts of these two systems. They used the Corine-plus land use classification system to differentiate between discrete organic and conventional agricultural intensities on the level of land use types. However, agricultural intensities may vary considerably within both conventional and organic production systems so this approach is not suitable to further differentiate between agricultural intensities within a specific agricultural production system.

Whether the method from Penman et al. (2010) would be suitable to differentiate between agriculture systems and

Table 3
Criteria for the classification of the suitability for a global application.

Study	Biogeographic coverage and resolution					Data availability	Our rating: global applicability
	Intended scope	Land use/land cover classification system	Scope of the land use/land cover classification system	Biogeo-graphic reference system	Scope of the biogeographic reference system	Use of global available biodiversity data	
Lindeijer (2000)	Global	None (Land use Situations, Test Cases)	Region specific (Europe, Scandinavia, South America)	Physiotope classification	Global	Yes	Middle
Brentrup et al. (2002)	Europe	Corine Land Cover	Europe	Biogeo-graphic regions	Europe	No	Middle
Kyläkorpi et al. (2005)	Global	Own Biotope Categories	Global	None	–	No	Middle
Koellner and Scholz (2007, 2008)	Europe	Corine Land Cover plus	Europe	None	–	No	Difficult
Burke et al. (2008)	Global	Biotope Categories after (Kyläkorpi et al., 2005)	Global	None	–	No	Middle
Michelsen (2008)	Global	None	–	WWF Ecoregions	Global	Partly (Key Factors)	Middle
Schmidt (2008)	Global	Own Land use categories	Global	None	–	No	Middle
Jeanneret et al. (2009, 2014)	National (Switzer-land)	Own Combination of land cover and use intensity	National	None	–	No	Difficult
Penman et al. (2010)	Global	None	–	None	–	(Yes) Expert opinions	Difficult
Geyer et al. (2010a, 2010b)	Region/San Joaquin Valley (USA)	Habitat Types (California Department of Fish and Wildlife)	California	Ecoregion (not further specified)	Global	No	Middle
Urban et al. (2012)	National	Corine Land Cover	Europe	None	Germany	No	Difficult
de Baan et al. (2013a)	Global	Classification after Koellner et al. (2013a)	Global	WWF Biome	Global	Partly	Middle
De Schryver and Goedkoop (2013)	Global	18 Land Use Types (ReCiPe Method)	Global	None	_	No	Middle
Mueller et al. (2013)	Global	Corine Land Cover Plus	Europe	WWF Biome	Global	No	Middle
de Souza et al. (2013)	America (North, Middle; South)	Classification after Koellner et al. (2013a)	Global	WWF Ecoregion	Global	No	Middle
de Baan et al. (2013b)	Global	5 broad land use types (LADA & Anthromes)	Global	WWF Ecoregion	Global	Yes	Easy
Coelho and Michelsen (2014)	Global (local)	Land Cover Database (LCDB2)	National (New Zealand)	WWF Ecoregion	Global	Partly	Middle
Lindner et al. (2014)	Global	None (Input Parameters)	_	WWF Ecoregion	Global	(Yes) Expert opinions	Middle
Saling et al. (2014)	Global	None (Input Parameters)	_	None	_	No	Difficult
de Baan et al. (2015)	Global/East Africa	63 land cover classes (GlobCover v2.3)	Global	Pixel (0.81 km^2)	Global	Yes	Easy
Chaudhary et al. (2015)	Global	6 land use types (LADA & Anthromes)	Global	WWF Ecoregion	Global	Yes	Easy

intensities depends on the experts and the questions on which the assessment would be based. We cannot appraise that in the current state of the method development, because no specific questions were formulated in this regard. The same limitation applies to the method of Lindner et al. (2014) which theoretically enables a differentiation between agricultural intensities but a precise means of implementation has not yet been found. All other methods are too coarse to differentiate between agriculture systems and intensities. The implication is that there are methods that can differentiate between agricultural production systems but these are restricted to a specific region or country (Jeanneret et al., 2009), and there are methods that are globally applicable but are not detailed enough to distinguish between agricultural systems or intensities (de Baan et al., 2013b; Chaudhary et al., 2015; de Baan et al., 2015).

This finding is supported by a FAO report which concludes that the existing biodiversity impact assessment methods are not suitable for comparing the biodiversity effects of different livestock production systems (LEAP, 2015). Anton et al. (2014) also mentioned that there are not enough impact characterization methods for a comparison of agricultural intensities. Therefore, the aim of further research and methodological development should be to resolve these existing shortcomings and to develop a method in which a differentiated biodiversity assessment of agricultural intensity levels is associated with global applicability. Development of impact assessment approaches for biodiversity in LCAs that enable a clear distinction between different agricultural systems would enable options for action for a sustainable use of biodiversity resources to be derived.

4. Conclusions

In this review, the land-use related life cycle impact assessment methods that have been developed and used in the last 15 years have been extensively analysed from the point of view of applying them in agricultural contexts. The analysis includes an overview of the chronological improvements and the current state of research, which has revealed strengths and weaknesses in the currently applied methodologies. A gap was identified between globally applicable methods, which are still too coarse to distinguish agricultural intensities, and methods which are able to distinguish agricultural intensities, but are restricted to a specific geographic range and have high data requirements. Despite the identification of meaningful improvements in the applied methodologies over time, we argue that more work needs to be done to develop appropriate globally applicable biodiversity assessment methods, with a resolution in which the biodiversity impact of various agricultural intensities can be differentiated. We suggest the value of bundling the strengths and learning from the progress of existing methods to pursue a consistent direction for further methodological developments.

A limitation of this review however, is that it remains a theoretical comparison of reported methods. Furthermore, most of the methods were not necessarily designed with the intention to be especially suitable for agricultural production. An approach to overcoming these limitations might be to apply the different methods in the same context in a comparative application study. Such an approach would be expected to gain insight in the scope of application of the commonly used methods, and serve a diagnostic purpose. Despite these limitations, the findings in this review allow us to make some recommendations that could be useful in guiding future research and methodological development.

The finding that many existing methods so far relate to the Land Use/Land Cover Types rather than to the conditions within the production biotopes themselves is analogous to viewing a picture frame rather than the picture. Important basics of the methods, such as the choice of biodiversity aspects, biodiversity indicators and reference conditions, should be reflected and compared with the current state of knowledge in biodiversity assessment in general. In particular, it should be critically questioned whether the derived conclusions agree with the initial assumptions, such as whether a selected indicator is really suitable for the evaluation of a specific biodiversity aspect. The reasoning for, and the informative value of, the choice of the biodiversity aspects, biodiversity indicators and reference conditions should be transparent and sound.

A further important recommendation is to take steps towards consistency in applied methodologies. We recommend for LCA assessments of agricultural products, that there should be default categories, including biodiversity, along with appropriate indicators that could be used consistently in LCAs. A list of important criteria for the selection of biodiversity indicators in respect to the applicability in LCAs would facilitate further method developments and would lead to more comprehensive biodiversity assessments. Future research efforts should be undertaken to develop appropriate measures, parameters, and indicators for the biodiversity assessment in LCAs. To enable biodiversity assessments of agricultural products in relation to different issues that are independent of the chosen reference condition, we propose the inclusion of a further step in the process: namely to compare the results of the biodiversity impact assessment of different agricultural production systems with the average land use practice, or an alternative use scenario, in the same region to show the biodiversity impacts of different agricultural use intensities.

The results allow the recommendation to use different assessment methods depending on the level of detail or the question referring to biodiversity, since not all questions require equally detailed assessments. A prerequisite for the selection and use of a particular method is reflection on the suitability of the methods for the specific demands and transparency in reporting. We propose to use a combination of at least one product related functional unit and an area related functional unit to assess the biodiversity impacts of agricultural production systems. The use of quality-related functional units (e.g. nutritional quality) could also give additional information. Landscape effects should also be included in the biodiversity assessment methods because agriculture shapes landscapes and the large influence of landscape attributes on biodiversity has been confirmed in many studies (Wetterrich and Haas, 1999).

The issue of refining globally applicable methods so that they can distinguish between agricultural intensities may be addressed by the inclusion of important biodiversity-influencing agricultural activities (e.g. nitrogen fertilization, pesticide use) into an existing, but coarse, biodiversity assessment method. This recommendation is, of course, dependent on the availability of data, which should be considered when selecting a suitable method. With this in mind, a guide for LCA-users to cross reference with which question and for which data availability, they can or should use a certain method would be a useful development. Such a guide would include a classification of the different available methods; from coarse to a high level of detail.

While the aim of this review might have been interpreted as a way of finding ‘best practice’ in biodiversity impact assessment in LCA, we rather see its value as identifying important starting points for further methodological development. In the future, numerous obstacles must certainly be overcome to develop globally applicable biodiversity impact methods for agricultural production systems. We acknowledge the work of the pioneers included in this review who have laid the foundation for effectively including biodiversity assessment in agricultural LCAs. The increased efforts, such as the Life Cycle Initiative, encourage the prospect of fulfillment of this hope.

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Evaluating On-Farm Biodiversity: A Comparison of Assessment Methods

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Abstract: Strategies to stop the loss of biodiversity in agriculture areas will be more successful if farmers have the means to understand changes in biodiversity on their farms and to assess the effectiveness of biodiversity promoting measures. There are several methods to assess on-farm biodiversity but it may be difficult to select the most appropriate method for a farmer's individual circumstances. This study aims to evaluate the usability and usefulness of four biodiversity assessment methods that are available to farmers in Switzerland. All four methods were applied to five case study farms, which were ranked according to the results. None of the methods were able to provide an exact statement on the current biodiversity status of the farms, but each method could provide an indication, or approximation, of one or more aspects of biodiversity. However, the results also showed that it is possible to generate different statements on the state of biodiversity on the same farms by using different biodiversity assessment methods. All methods showed strengths and weaknesses so, when choosing a method, the purpose of the biodiversity assessment should be kept in the foreground and the limitations of the chosen methods should be considered when interpreting the outcomes.

Keywords: biodiversity assessment methods; agriculture; case study; on-farm evaluation

1. Introduction

There is little dispute that agriculture is one of the main causes of global biodiversity loss [1] and that agricultural induced biodiversity decline is accelerated by intensification and expansion of agricultural land use [2,3]. Several of the factors that can be considered as intensification, such as increased drainage, irrigation, grazing intensity [4], and the homogenization of landscapes, lead to the loss of habitat elements. Other intensification factors, such as the increased use of fertilizers, herbicides, and pesticides, lead directly or indirectly to the death of organisms and their subsequent disappearance from the landscape [5]. Biodiversity loss in agricultural landscapes was highlighted in an alarming study that showed a 75% decrease in insect biomass in Germany over a period of just 27 years [6]. However, agricultural ecosystems are themselves dependent on various ecosystem services, such as water regulation, soil fertility, pest and disease control, and pollination [7]. Ollerton et al. [8] estimate, for example, that 87.5% of all flowering plants are in need of pollination by animals. Gallai et al. [9] estimated the economic value of the ecosystem service: "pollination" alone at around 153 billion euros, which corresponds to almost 10% of global agricultural production.

A wide body of research suggests that agricultural landscapes actually have the potential to contribute to biodiversity preservation [10] and can self-provide important ecosystem services when diverse agricultural systems are maintained [11]. Therefore, it is clear that the biodiversity loss in agricultural landscapes, which contributes to global biodiversity loss and the associated reduction of ecosystem services, is a result of insufficient diversity in agricultural systems, or over-intensive agricultural management, rather than by agriculture per se [12]. Consequently, governments recognize a global priority to reduce biodiversity loss [13] and agricultural management is often the focus of political activities that attempt to counteract this loss.

Efforts to stop biodiversity loss in agricultural landscapes can be enhanced by on-site engagement by farmers [14]. With approximately 40% of land in the European Union under agricultural use [15], farmers have a great potential to protect or even promote biodiversity with their management decisions, such as the mode of production they choose for their farms [14]. However, farmers need sufficient knowledge to make informed decisions, which implies a need for assessment methods to evaluate and monitor the biodiversity in their particular circumstances [10]. Stöckli et al. [16] recommended using such methods to support farm advisory services.

Farmers need both access to adequate information and the skills to process it if they are to make informed decisions about goal-oriented land management [10]. However, farmer education rarely provides farmers with the requisite skills and knowledge to achieve specific environmental outcomes by appropriate land management [17], so they typically consult with expert advisors. Gabel et al. [18] concluded that advisory services are an effective means of conveying biodiversity information to farmers, which reflects the findings of Noe et al. [19] who reported that dialogue with trained biologists was able to influence farmers' perceptions and awareness of wildlife. Similarly, Chevillat et al. [20] found that proposed on-farm biodiversity advice at a whole farm level, when delivered by a trained and credible advisor, raised the willingness of farmers to implement biodiversity conservation measures.

However, passing the task of biodiversity assessment at farm level to advisors is not without drawbacks. Smallshire et al. [21], while acknowledging the effectiveness of advice, pointed out that direct on-farm advice by well-trained advisors is also the most expensive way of disseminating information to farmers. There are also potential conflicts of interest in which advisors are sometimes incentivized to help farmers maximize their agri-environmental grants rather than providing them with the tools to undertake actions with the most environmental benefit [22]. Schroeder et al. [23] found that advisors tended to take a relatively neutral stance on on-farm biodiversity, which meant they experienced limited success in motivating farmers to implement biodiversity conservation measures. Ironically, farmers tend to trust advisory services that are perceived to be pro-farmer, and who have knowledge of agriculture and agri-environmental grants. In any case, an economical and readily applicable method for objectively evaluating the state of biodiversity on a particular farm could provide a useful start to discussions about on-farm biodiversity and may thereby contribute to motivating farmers to engage.

Jenny et al. [24] pointed out that it was difficult for farmers to quantify on-farm biodiversity, which limits their ability to evaluate the outcomes of biodiversity conservation efforts on their farm. Even for trained ecologists, it remains a challenge to assess or measure biodiversity or biodiversity loss [25] because biodiversity encompasses different biotic levels: from genes to ecosystems [26], and cannot be measured as a whole [27,28]. Duelli and Obrist [29] summarized that no single indicator for biodiversity can exist so appropriate surrogates must be used [30].

A possibility to help farmers recognize biodiversity changes could be provided by biodiversity assessment methods that can be applied on farms and thereby demonstrate the biodiversity impact of the farm or different production methods. Several methods have been developed to evaluate and monitor biodiversity [31–36], but there are fewer methods that can assess farmland biodiversity at a farm level [37]. Of those that can, there has been little comparative study into their usability, comparability, or accuracy, and the usefulness of these methods for individual farmers to gain a picture of their biodiversity conservation performance has not been extensively tested in real-world situations. Furthermore, many of these methods capture only part of the complexity of biodiversity [10] and it has not been sufficiently evaluated whether the results provided are valid with respect to the assessed biodiversity aspect. Primary research, in which these newly developed methods are applied to case study farms so that their outcomes and usability can be compared, could shed some light on the usefulness of these methods.

The aim of this research is to evaluate and compare a selection of biodiversity assessment methods by applying several methods to each of a set of case study farms. This approach, along with a measurement-based biodiversity evaluation on the same farms, will inform about the accuracy and suitability of these methods for assessing farmland biodiversity at the farm level. The methods that were applied, and the justification for selecting a case study approach, are described in detail in the methodology section.

2. Materials and Methods

2.1. Biodiversity Assessment Methods

To assess farmland biodiversity at farm level, four different methods were compared: (1) the credit point system (CPS) [31], which is a scoring system integrating various measures to promote biodiversity; (2) The Sustainability Monitoring and Assessment Routine (SMART), which is a comprehensive sustainability assessment tool that covers biodiversity as one thematic area [34]; (3) the Life Cycle Impact Assessment (LCIA) method for farmland biodiversity [32], in which species diversity on landscape scale is assessed as a potential loss due to land use intensity and deficiency of landscape structural elements; and (4) a traditional method of species monitoring in which numbers of individuals are determined and classified [31]. These four methods can be readily applied in Switzerland because data is either available or readily collectable. However, although these methods do not produce sufficiently similar values that allow them to be directly compared, they all evaluate farmland biodiversity and allow within-method farm comparison. The capacity for within-method comparison means that farms in which the same method has been applied can be easily ranked from best to worst.

(1) Credit Points System [31]

Within the project "Scoring with biodiversity—farmers enrich nature", the Research Institute of Organic Agriculture (FiBL) in Switzerland and the Swiss Ornithological Institute developed a scoring system to assess species diversity of agricultural farms. Relevant measures, from a list with 32 different options, most of which derive from the Swiss agri-environment scheme, for improving biodiversity at the farm level, are evaluated and recorded, which results in an overall score per farm. A particularly important component of the point system is the percentage of ecological compensation areas (ECA), which are mainly semi-natural habitats, on the farm. The "ecological quality", the size and the spatial distribution of those ECAs are also important [30]. Different in-field options for arable land and grassland, and farm characteristics, such as average field size, are also included in the assessment [30]. The points are weighted with the help of expert judgements and on the basis of the known benefits for biodiversity and thus, for instance, larger meadows receive a higher score than smaller meadows and those with a high ecological value receive more points than meadows with a lower ecological value [24].

(2) Sustainability Monitoring and Assessment Routine [34]

The Sustainability Monitoring and Assessment Routine (SMART) is a method that was developed by FiBL to assess the sustainability performance of farms in a comprehensive, efficient and comparable way. The tool is based on the Sustainability Assessment of Food and Agriculture systems (SAFA) Guidelines of the FAO [38]. SMART covers four sustainability dimensions (good governance, environmental integrity, economic resilience and social well-being) which in turn consist of 21 themes and 58 subthemes. Biodiversity is reflected as one theme that encompasses the subthemes: genetic diversity, species diversity, and ecosystem diversity. The SMART method is built on a set that includes up to 327 indicators. A total of 72 indicators are related to the biodiversity theme [34] and are primarily concerned with possible management measures and factors relating to use intensity. For each indicator, the achievement of the goal is measured in a range of 0–100%. The method used in this study was SMART-farm tool version 4.1.

(3) Life Cycle Impact Assessment [32]

The Life Cycle Impact Assessment (LCIA) method of Meier et al. [32] was developed to assess agricultural land use related impacts on farmland biodiversity. It allows for the continuous assessment of the effects of various agricultural production intensities (e.g., organic and conventional production) on biodiversity and includes the local landscape context. For brevity, and for the purposes of this paper, we refer to the LCIA biodiversity assessment method as the LCIA method or simply LCIA.

The LCIA method is based on regression models that describe species diversity on landscape level as a function of land use intensity and landscape structure including parameters that refer not only to the farm under consideration but also to the surrounding landscape. The regression functions are based on empirical data [39] and are valid for European agricultural landscapes in the lowlands. The species groups included vascular plants and birds, and the landscape level refers to a square of 4×4 km. The model first assesses the impact on the diversity of the mentioned species groups on landscape level by considering the nitrogen input on the agriculturally used area; the diversity of the crop rotations within the overall landscape; and the proportion of semi-natural area, with individual elements converted to an equivalent area value, within a specific landscape. This determines the overall impact in a landscape due to agricultural land use. The overall impact at the landscape level is then allocated to a specific farm within the landscape proportionally to the land use intensity and the proportion of semi-natural habitats on that farm. Semi-natural habitats cover aerial structural elements, such as permanent grassland, forest, fallows, bogs; and linear structural elements, such as hedges, tree lines, grassy margins, and solitary trees. The impact on farmland biodiversity is expressed as the biodiversity damage potential (BDP), which is calculated by comparing the species diversity, under the given land use intensity and landscape structure, with the species density that would be expected under a state of minimum land use intensity and 100% semi-natural habitats. The BDP is normalized to a dimensionless index with values between zero and one in which lower values indicate land use that is better for encouraging species diversity.

(4) Monitoring plant and species diversity [31]

Biodiversity data were collected during several visits on the five farms throughout the years 2009 and 2015. On each farm, transects with a total length of 2500 m were laid, in which the diversity (among other things, species number and density) of plants, grasshoppers, butterflies and birds were recorded. The transects were distributed over the farm so that all ECAs and arable land or grassland types were covered and the total number of all species on the farm was counted to calculate the species richness. Species richness simply quantifies how many different species were contained in the dataset. Calculation of other commonly used indices of species richness, such as the Shannon Index, Simpson Index, and Berger-Parker index was deemed unnecessary for the purposes of this study However, species richness is a function of farm size and the farms in this study were not identical in size, so a density measure, which is independent of the farm size, was calculated by summing all individuals of each transect and dividing by the farm area. The entire farmland was used for the bird recordings and all birds, which were heard or seen, have been incorporated. For a precise description of the measurement-based evaluation of biodiversity, see Birrer et al. [31].

2.2. Case Study Approach

According to Yin [40] a case study is an empirical study that examines a phenomenon in a real-life context: especially when the boundaries between the phenomenon and the context are not entirely clear. A central component of case studies, which this study also aims to pursue, is to attract attention and make suggestions for specific relations [40]. The execution of several case studies should help to support either similar or, for predictable reasons, contrasting results [40]. It is not the aim of case studies to derive generalizations [41].

We contrasted the four methods on each of the five farms to reveal the advantages and disadvantages in their application as well as their accuracy. To this end, each method was compared with the measurement-based evaluation of biodiversity. Although indicator-based methods have been developed with the explicit aim of evaluating biodiversity with less time and fewer resources than measurement-based methods, both measurement based and indicator-based biodiversity evaluation methods are quite time- and resource-intensive to apply. These time and resource demands meant that this comparison study, in which four methods were applied to each case study farm, was restricted to five farms. Furthermore, remembering that the LCIA method requires calculation of a landscape structural input parameter, so the biodiversity damage potential (BDP) is usually calculated for the entire farm within a 4 × 4 km quadrant. In our case, we extended the area to 5 × 5 km landscape quadrants to make sure that all areas of the farms could be included. Extending the size of the quadrants was not considered to negatively impact the logic of the method, with the BDP expressing the influence on biodiversity of the entire farm in the considered landscape. Thus, the results depend strongly on the respective farm size, with larger farms having a greater influence on the BDP. This applies to a positive as well as a negative influence on biodiversity.

2.3. Management Variety on Swiss Farms

Essentially, there are three production systems in Switzerland to which virtually all Swiss farms belong. Swiss farmers who meet a number of prescribed ecological standards [42] and thereby qualify for general direct payments are known as proof of ecological performance (PEP) farms. Organic farming is additionally subsidised, and Swiss law dictates the minimum requirements, including implementation of at least 12 from a catalogue of ecological measures, that a farm must meet to gain the organic label and claim further direct payments. A parallel label in Switzerland is IP SUISSE, which is a joint NGO/private initiative for integrated pest management, and which is based around a system of points being allocated to implemented measures, from a pre-defined list, that demonstrate the ecological performance of the farm. Farmers who achieve sufficient points gain the IP SUISSE accreditation label and qualify to receive higher product prices from some major retailers. The IP Suisse label was founded to reward farmers for providing additional services, such as limited use of chemical sprays and fertilizers, which remain at the heart of the IP SUISSE philosophy. Although the use of chemical sprays and fertilizers is among the practices that are forbidden in organic production, both organic and IP-SUISSE-labelled farms have some freedom to decide which measures they implement to meet standards or gain sufficient points to attain their respective label [37].

Therefore, it can be assumed that the production intensity of an organic or IP Suisse farming system will be less intensive than a conventional PEP farming system. Gomiero et al. [43] noted that the biodiversity potential of organic farms is greater compared with conventional farms through a higher habitat variability, more wild-life friendly farm practices and, although less decisive, by the renunciation of pesticides. Nevertheless, the production intensities within a production system can also vary widely, with Weibull and Östman [44] and Billeter et al. [39] finding that biodiversity in agricultural ecosystems depends on both landscape heterogeneity and farm management methods.

2.4. Sampling

Remembering that the aim is not to compare production systems, we used a maximum variety sampling strategy [45] to select farms with different management intensities from within the different production systems to provide contrasting environments and thereby facilitate comparisons of the biodiversity assessment methods. Duelli et al. [46] assigned a higher organismal biodiversity to habitats that are less intensively cultivated, so a prerequisite for the selection of the case study farms was that they should cover a large variety in management intensity and represent the gradient found within different production systems.

The farms were selected so as to be located sufficiently close to each other, in the cantons Bern (two farms) and Lucerne (three farms), that the LCIA landscape structural input parameter only needed to be calculated once for each region. The distance between both landscapes was approximately 70 km but both sites belong to the Swiss lowlands at an altitude of 543–803 m above sea level (Figure 1). An average precipitation probability of 1100 mm is assumed for all farms. A total of 16.9 ha are farmed on the smallest farm and 30.2 ha on the largest farm. Further information can be found in Table 1.

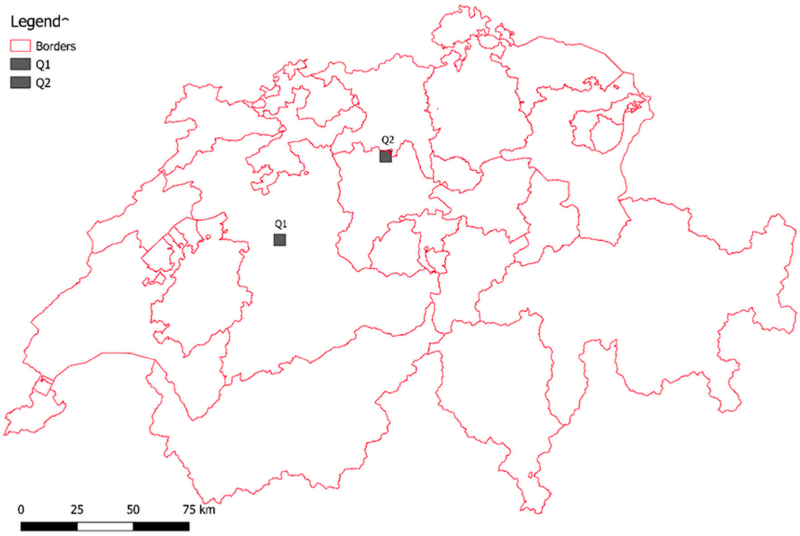

Figure 1. Location of the landscape squares in Switzerland (Q1 in canton Bern, Q2 canton Lucerne).

All farms are mixed farms. Originally, all five farms had dairy cows whereby one farm stopped keeping dairy cows in 2015. In addition, pigs, sheep or horses were kept on some farms. The stocking density on the farms ranged between 0.8 and 3.16 livestock per hectare. The forested area on the farms ranged between 0.08 ha and 4.98 ha. Between four and seven different crops are cultivated within the crop rotation. All farms exceed the 7% ecological compensation area required in Switzerland with the proportion of ECAs ranging between 10.3% and 13.9%.

Table 1. Overview of farm details.

Farm	Farm 1	Farm 2	Farm 3	Farm 4	Farm 5
Landscape Canton (municipality)	Bern (Rubigen)	Lucerne (Rickenbach)	Bern (Rubigen)	Lucerne (Rickenbach)	Lucerne (Rickenbach)
Production system	IP Suisse	IP Suisse	Demeter	Bio Suisse	PEP
Altitude (m)	543	803	586	702	672
Precipitation (mm)	1100	1100	1100	1100	1100
Agricutural area (ha)	36.9	27.2	23.5	23.4	16.9
Arable land (ha)	30.20	9.8	15	17.48	9.09
Pasture (ha)	5.85	16.38	7	5.94	3.9
Forest (ha)	0.08	4.98	2.18	3.6	3.13
Animals	Cows/sheep	Cows/pigs	Cows, pigs, horses	Cows/pigs	Cows/Horses
Livestock units per ha	0.8	2.08	1.7	3.16	1.59
Crops	Wheat, Barley, rye, rapeseed, sugar beet, peas, sunflower, maize for silage	Barley spelt wheat	Broad Beans, Oats, Wheat, spelt, peas, maize for silage	Sunflowers, potatoes, maize for silage, wheat, spelt	Wheat, rapeseed, barley, maize for silage
Crops in rotation	7	4	7	6	5
Labour force (Full-time equivalent)	2.5	2.8	4	3	1.45
ECA (ha)	3.8 ha	3.78	3.20 ha	2.6 ha	1.84 ha
%	10.3%	13.9%	13.6%	11.1%	10.6%

2.5. Analysis Criteria

To examine the usability of the different biodiversity assessment methods, a set of criteria was established which describes the basis of the methods and can provide guidance in selecting the appropriate methods for the different purposes. The selection of the analysis criteria was based on criteria that have been used in an overview of biodiversity assessment methods for LCIAs [47] or in an analysis of sustainability assessment methods in food systems [48]. The following analysis criteria were selected:

- The biodiversity aspect describes the part of biodiversity that is covered by the various methods (e.g., species biodiversity, genetic diversity or functional diversity).
- Since it is not possible to measure biodiversity as a whole [49], the methods depend on the use of biodiversity indicators. Biodiversity indicators can be samples (e.g., plant richness) or indirect measures (e.g., semi-natural habitats), which provide information to express the size, extent or degree of biodiversity [50].
- To compare biodiversity, it is necessary to use a benchmark. This is referred to as the reference situation for the investigated methods.
- The spatial unit refers to the reference value on which the biodiversity assessment is based on (e.g., product, plot, field, farm).
- Geographical scope refers to the geographical context in which the methods can be applied. It can be distinguished, for example, whether the methods can be applied regionally, nationally or globally.
- Primary purpose means the originally intended use of the method. Examples of this are research, advice, evaluation, etc.
- The duration of the assessment describes the time required to generate a result with the methods.
- Data availability refers to the sources of the data that are required for the application of the methods.

A comparison of different methods is difficult because of the different ways they describe biodiversity. For the CPS method, the system focusses solely on species diversity, while the biodiversity theme in the SMART method is subdivided into species diversity, ecosystem diversity and genetic diversity. The LCIA method covers the biodiversity aspect: 'species richness' on a landscape scale and also indirectly covers ecosystem diversity. All methods are indicator based.

Furthermore, the reference point in each of the methods is different. The CPS compares with a reference point of 0 points, the SMART method refers to a 100% percent goal achievement, and the LCIA method refers to a potential maximum number of species. Despite different reference points, one way of comparison is to rank the farms according to the outputs from the different methods.

3. Results

3.1. Overview of the Analysis Criteria

These criteria provide useful guidance when selecting the most appropriate method to choose in specific circumstances. In particular, they indicate the relationship between the selected method and the purpose for which the evaluation is being conducted. The results also indicate the resource commitment required for them to be applied, which may be decisive in many situations. Furthermore, it can be seen that each method requires a degree of expertise and so may be beyond the ability of farmers to implement self-assessment on their farms.

3.2. SMART

The SMART method provides an overall score for the thematic area of biodiversity, which is expressed as a percentage of the goal achievement. The percentage is calculated from the average of the results for the subthemes ecosystem diversity, species diversity and genetic diversity (Table 2).

Table 2. Results of the Analysis Criteria.

Method	CPS	SMART (Version 4.1)	LCIA	Measurement
Biodiversity aspect	Species diversity	Species diversity, ecosystem diversity genetic diversity,	Species diversity (indirect Ecosystem quality)	Species richness, species density
Biodiversity indicator	32 measures options to enhance biodiversity on the farm (most of them base on options of the Swiss agri-environment scheme. Management measures)	72 biodiversity indicators	Land use intensity parameters (Land use intensity index N-input, and crop diversity) and landscape structure parameters (Share of semi natural habitats)	Number of individuals, identified to species level
Reference situation	0 Points (although 0 points is extremely unlikely)	100% goal achievement	maximum possible number of species	None. It is not comparative
Spatial Unit	Area (Farm)	Area (Farm)	Area (Farm), product	Area (Farm) (can be calculated per hectare)
Geographical scope	Switzerland	Global	Primarily temperate regions (lowlands)	Global
Intended Purpose	Biodiversity Assessment and Advice on farm-level	Sustainability assessment for research and business contexts	Assessment of farmland biodiversity of agricultural products in the framework of a LCIA	Biodiversity assessment at a farm level
Assessment duration	2 h	3–4 h, but including an evaluation of other sustainability indicators other than biodiversity.	Depends strongly on the availability of processed data (especially the landscape data)	A half day, 3–6 times per season, with approximately one day in the laboratory per animal/plant type
Data availability	On the farm available	On the farm available	Partly on the farm available, additional landscape data needed	On the farm
Expertise needed	High expertise in biology and knowledge of subsidy system	Moderate expertise. Lay people require some training	High expertise in GIS	High expertise in biology

The results of the farms for the biodiversity theme ranged from 46 to 66% (shown in Table 3). The highest and lowest values were achieved by farms 3 and 1. Farms 4 and 2 were ranked second and third respectively, while farm 5 was ranked fourth. Farms 1 and 3 retain their position in the ranking for all three sub-themes. On the other hand, the results of the other three farms at the subtheme level did not remain constant, but were quite different for the individual sub-themes. The theme values were used in the comparison with the other biodiversity assessment methods because they give the broadest assessment of biodiversity.

Table 3. Sustainability Monitoring and Assessment Routine (SMART) theme and subtheme results.

Farm	Farm 1	Farm 2	Farm 3	Farm 4	Farm 5
Production system	IP Suisse	IP Suisse	Organic (demeter)	Organic	PEP
Farm size (ha)	36.9	27.2	23.5	23.4	16.9
Theme Biodiversity	46%	56%	66%	61%	55%
Subtheme Ecosystem diversity	43%	50%	61%	56%	57%
Subtheme Species Diversity	48%	57%	70%	70%	54%
Subtheme Genetic Diversity	47%	60%	68%	57%	55%

3.3. CPS Method

The CPS scoring system for the five different farms results in scores between 13.4 and 26.8 points. Farms 3 and 2 were ranked highest with scores with 26.8 and 26.7 points respectively. Farm 4 was ranked third with 18.8 points and farm 5 was ranked fourth with 15.4 points. Farm 1 was ranked worst with 13.4 points.

3.4. LCIA Method

The results of the assessed farms using the LCIA method reflect the relationship between farm size and biodiversity impact (Table 4). The BDP of farm 1, for example, is the largest, since it also clearly has the largest farm area. Accordingly, the smallest farm 5 has the least negative impact on biodiversity and its size is less than half that of the largest farm. Farm 2 can also be grouped according to the farm size. There is a difference from the ranking according to the farm size only for farms 3 and 4, which are similar in size, with the results showing that the slightly larger farm 3 has a lower BDP than the slightly smaller farm 4. These reversed places in the ranking can be explained by the comparably lower intensity of farm 3 and the very small difference in size (0.1 ha) between the two farms. For the outcomes of this method to give more than an indication of farm size, it was necessary to calculate the BDP per hectare and these results are shown in Table 4. The ranking according to BDP per hectare shows that farm 3 has lowest BDP per ha, which was followed by farm 1, which was ranked second. Farm 4 was ranked third in both BDP and BDP/hectare, while farm 5 was ranked fourth. The fifth place in the BDP ranking was farm 2, which has the greatest negative impact on biodiversity in the landscape per hectare. The BDP/hectare values were used in the comparison with the other biodiversity assessment methods because they are independent of the farm area.

Table 4. Results of the Life Cycle Impact Assessment (LCIA) biodiversity damage potential on farm.

	Farm 1	Farm 2	Farm 3	Farm 4	Farm 5
Production system	IP Suisse	IP Suisse	Organic (demeter)	Organic (Biosuisse)	PEP
Farm size (ha)	36.9	27.2	23.5	23.4	16.9
LCIA BDP/farm	7.3×10^{-3}	6.6×10^{-3}	4.4×10^{-3}	4.7×10^{-3}	3.5×10^{-3}
LCIA BDP/ha	1.97×10^{-6}	2.43×10^{-6}	1.86×10^{-6}	2.01×10^{-6}	2.06×10^{-6}

3.5. Monitoring Plant and Animal Biodiversity

In Table 5, the rankings of the individual numbers for species richness and density have been added to form an overall ranking for both measures. While the largest farm in the species richness ranking scored relatively well to rank second, this changes within the density ranking where the farm has the worst rank. The smallest farm, which clearly scored the worst in the species richness ranking, was ranked fourth in species density, while the second largest farm 2 was ranked first in both rankings, although in the density ranking, this place was shared with farm 3. Farm 4 was ranked third in both rankings. The density values were used in the comparison with the other biodiversity assessment methods because they are independent of the farm area.

Table 5. Results of the biodiversity recordings on the farm 2015.

Farm	Farm 1	Farm 2	Farm 3	Farm 4	Farm 5
Production system	IP Suisse	IP Suisse	Organic (demeter)	Organic (Biosuisse)	PEP
Farm size	36.9	27.2	23.5	23.4	16.9
Species richness Butterflies	22	26	24	26	18
Species richness plants	205	185	160	166	129
Species richness birds	35	25	24	23	21
Species richness grasshoppers	8	13	7	6	7
Ranking species richness	8	6	13	13	18
Density plants	695.579	925.723	1051.633	791.437	596.752
Density Grasshoppers	82.441	229.363	163.416	88.982	95.773
Density Butterflies	60.45	178.859	364.894	382.726	203.576
Density Birds UZL	3.25	2.205	0.852	0.427	1.179
Ranking Density	15	9	9	13	14

Note: Species richness refers to the number of species per farm. Density refers to the number of individuals per ha.

3.6. Comparison of Rankings According to Evaluation System

The combined results are shown in Table 6.

Table 6. Overview of the different method results (ranking in brackets).

Farm	Production Systems	Farm Size (ha)	CPS Biodiversity (Points)	Smart Theme Biodiversity (Goal Achievement)	LCIA Methode (BDP/ha)	Species Density (Individuals/ha)
Farm 1	IP Suisse	36.9 (1)	13.41 (5)	46% (5)	1.97×10^{-6} (2)	15 (5)
Farm 2	IP Suisse	27.2 (2)	26.71 (2)	56% (3)	2.43×10^{-6} (5)	9 (1)
Farm 3	Organic (demeter)	23.5 (3)	26.77 (1)	66% (1)	1.86×10^{-6} (1)	9 (1)
Farm 4	Organic (Biosuisse)	23.4 (4)	18.79 (3)	61% (2)	2.01×10^{-6} (3)	12 (3)
Farm 5	PEP	16.9 (5)	15.37 (4)	55% (4)	2.06×10^{-6} (4)	14 (4)

A comparison of the three biodiversity assessment methods with the species recording data of the farms shows that the CPS scoring method corresponds reasonably closely to the density data. There is the restriction that in the species density ranking results the first place was given twice and farm 2 and 3 are thus equal and there is, therefore, no second place in this case. There is also agreement between the rankings of the LCIA method and the density values for the farms 3, 4 and 5, which occupy the first, third and fourth places respectively. The farms 1 and 2 show the same deviations as for the CPS scoring method. The SMART ranking is also very similar to the density ranking and is in agreement with it on the ranks 1, 4 and 5. Farm 4 scored in the SMART rating: one place better than in the recording data, and ends on the second place. The highest deviation from the ranking with the density values can be found for farm 2, which comes third in the SMART method after the two organic farms 3 and 4.

The best evaluation in all four methods is given to the farm 3. With one exception, the ranks obtained with the CPS scoring method and the SMART method matched. In the CPS evaluation, farm 4 was ranked third with farm 2 ranked second, while the SMART method ranked farm 4 scores slightly better in second place with farm 2 in 3rd place. (The subtopic genetic diversity in the SMART method shows exactly the same order of the farms as the CPS score). For the LCIA method with BDP/ha, the ranking corresponds to the CPS method in the farms ranked first, third and fourth. However, in contrast to the other two methods, farm 1 was ranked second rather than fifth, while farm 2 was ranked fifth.

4. Discussion

Compared to the other two methods, relatively few input parameters are decisive for the evaluation of farms with the LCIA method. Possibly this is also the reason that this method showed a different picture of biodiversity than the other methods for two of the five farms. It could be that crop diversity plays a too large a role in the LCIA method because the two farms that showed a clear deviation from the rankings of the other farms, differ most from each other. While farm 1, as a seed producing farm, has up to 7 crops in its crop rotation, only 4 different crops are cultivated on farm 2. This might also be the case for the animal stocking density, where farm 1 performs clearly better than the other farms (first place with farm 4 in 4th place). It may, however, be that differences occur because the LCIA method applies a model that focusses on species diversity at the landscape level in which intensity and landscape structure in the remaining landscape are also considered. Further research with a larger sample would be needed to explore this possibility.

When comparing the SMART method with the CPS method, there is only one difference in the ranking of the two: organic farm 4 is rated one place better (second place) with the SMART method than with the CPS method. This means that both organic farms are ranked best with SMART. One reason for this, which has already emerged in other comparative studies between organic farms and conventional farms using the SMART method, is that possibly too much importance is attached to the use of synthetic pesticides (which are forbidden on organic farms). In addition, in the SMART method so far only the type and number of pesticides used have played a role, but not the amount and number of sprays used on the farm. In this case, farm 2, which uses pesticides sparingly, was probably overly penalized. The clear advantages of the SMART method, however, are the comparatively low workload, the global applicability and the fact that, in addition to the biodiversity result, many other sustainability issues are also covered by the method. Although Smallshire et al. [21] pointed out that direct on-farm advice by well-trained advisors is also the most expensive way of disseminating information to farmers, none of the methods tested in this study are readily implemented without a reasonably high degree of expertise, so advisors are necessary for their implementation.

There were some limitations of the study: the most important of which was the small sample size that was necessitated by conducting four resource intensive methods of assessing biodiversity on the case study farms. Further study with a larger sample would be recommended to confirm the results of this exploration. Sample size notwithstanding, this study provided validation for each evaluation method in some aspects of biodiversity, and could show the strengths and weaknesses of the selected methods. With five evaluated farms, we cannot make statistical statements about significant differences and we are aware of Hays' [41] caution that a case study design is not intended to meet in generalizations.

The approach of to using a ranking system to compare farms is also potentially problematic in that the difference between the ranking was sometimes very small. For example, the difference between the second (1.97×10^{-6}) and fourth (2.06×10^{-6}) ranked farms according to the LCIA method was 0.000006. Indeed, most farmers might be challenged to interpret, or qualify, these numbers. However, despite these challenges, ranking is how biodiversity assessment is typically carried out in the real world. This can be a ranking of a particular set of farms, or an indication of change with an individual farm in which biodiversity is evaluated at different points in time. As the results of this study have shown, it is quite possible to generate different statements about biodiversity by applying different methods to the same farms. We argue that the approach of ranking avoids giving a false picture by maintaining a methodological consistency, and allows qualification of the results by means of comparison.

Buchs [49] and Duelli and Obrist [46] each pointed out that biodiversity, in its entirety and complexity, is not readily measurable and there can be no all-encompassing indices that could be derived for it. These statements were supported by the results of this case study in which no method was able to make a definitive statement about the actual state of biodiversity on the different farms. For example, even the method in which species richness was measured on site by trained ecologists, there was variation between farms, with the farm that had the highest density of birds also found with the lowest density of grasshoppers and butterflies. Furthermore, for the attempt to determine the accuracy of the methods, we made a comparison of the methods with the species recordings on the farm, but it has to be clear that the species recordings also only represent one specific aspect of biodiversity. In this case, the measurement method represents the species density aspect for a few indicator species (birds, plants, butterflies and grasshoppers). We do not have any indication for genetic and ecosystem diversity on the farms, which Purvis and Hector [26] suggest would be necessary for an evaluation of biodiversity. However, these methods can each be understood as indications or approximations to this state or at least for one or more aspects of the state and thereby deliver the information demanded by Clergue et al. [10]. If the possibilities for making statements are clearly defined or limited from the beginning, and the appropriate analysis criteria for the particular circumstances are considered, they can be helpful in deciding which method is most suitable.

5. Conclusions

When selecting a method to apply, and given that the methods compared within this study all exhibit strengths and weaknesses, with no method producing results that are overwhelming, a farmer might be advised to choose the evaluation method that is the least effort. The CPS method was developed especially for the Swiss context (even especially for lowland farms) and mainly integrates Swiss agri-environmental measures. It appears to provide an accurate picture of the species diversity on Swiss farms, but is not globally applicable and therefore might not allow a comparison of farms from different regions of the world. It is only suitable as a self-evaluation tool for farmers and advisory support in Switzerland if they have a sufficient level of expertise in biology and knowledge of the Swiss subsidy system.

The SMART method was found to be the least resource intensive and requires the least expertise while also producing a reasonably accurate picture of the on-farm biodiversity. The SMART method would also be preferred if the farmer saw biodiversity within the frame of a more complex view of sustainability, because biodiversity is one of several sustainability aspects considered within the SMART tool. The LCIA method appears to be more suited to an evaluation of biodiversity that is connected to a product comparison or in cases where it is not possible to visit the farm in person. Traditional measurement by counting individuals has a high degree of credibility, but is resource intensive, requires a high level of expertise, and is time-consuming. This method appears to be more suitable for confirming the accuracy of the faster and cheaper indicator-based methods. In summary, none of the methods is capable of giving a clearly better estimate of actual on-farm biodiversity than any other. Choice of method would best be made in light of the purpose of the evaluation.

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KANN BIODIVERSITÄTSBERATUNG DIE EINSTELLUNG ZUR BIODIVERSITÄT VON SCHWEIZER LANDWIRTEN UND LANDWIRTINNEN BEEINFLUSSEN?

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Den Landwirten und Landwirtinnen kommt eine Schlüsselrolle bei der Erreichung der schweizerischen Biodiversitätsschutzziele zu. Aber wie können sie motiviert werden, sich mit diesen Zielen zu identifizieren? In einer sozialwissenschaftlichen Studie untersuchten das Forschungsinstitut für biologischen Landbau FiBL und die Schweizerische Vogelwarte gemeinsam den Einfluss von gezielter Biodiversitätsberatung auf die Einstellung und Motivationen von Schweizer Landwirt/innen. Die Ergebnisse liefern wirkungsvolle Ansatzpunkte für die Überzeugungsarbeit von Biodiversitätsberater/innen.

Einleitung

Rund ein Drittel der Schweizerischen Landesfläche wird landwirtschaftlich genutzt und immer intensiver bewirtschaftet. Die Intensivierung äussert sich beispielsweise durch den Einsatz von grossen und leistungsstarken Maschinen, hohem Viehbesatz, Kraftfutterimporten oder hohen Nährstoffgaben, aber auch durch das Verschwinden naturnaher Landschaftselemente. Diese Entwicklung trägt zu einer Abnahme der Artenvielfalt in den landwirtschaftlichen Gebieten bei. Als Reaktion auf den Artenverlust müssen Landwirt/innen seit 1993 als Kompensationsmassnahme mindestens 7 % der landwirtschaftlichen Nutzfläche als Biodiversitätsförderflächen (BFF) bewirtschaften und bekommen Direktzahlungen dafür. Trotz der stetigen Weiterentwicklung dieser ökologisch motivierten Direktzahlungen nimmt die Biodiversität in der Schweiz nach wie vor ab. Eine mögliche staatliche Massnahme wäre es, die Gesetzesbestimmungen zu verschärfen, z.B. indem ein höherer Anteil von BFF im Ackerland verlangt oder der maximale Abstand zur nächsten BFF vorgeschrieben würde. Ein solcher Ansatz stiesse aber auf grossen Widerstand, weil den Landwirt/innen ihre Autonomie sehr wichtig ist. Erfolgsversprechender sind daher Ansätze, bei denen Landwirt/innen von der Bedeutung der Biodiversitätsfördermassnahmen und dem ökonomischen Nutzen für die Produktion und den Betrieb überzeugt werden. Das heisst, den Schutz der biologischen Vielfalt in den Herzen und Köpfen der Landwirt/innen zu verankern. Es bleibt aber bisher noch unklar, wie die Landwirt/innen am besten motiviert werden können der Biodiversität einen höheren Stellenwert einzuräumen. In dieser Studie wurden drei Ziele verfolgt: Das erste Ziel war,

LA VULGARISATION AGRICOLE PEUT-ELLE INFLUENCER LE RAPPORT DES AGRICULTEURS SUISSES À LA BIODIVERSITÉ?

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En Suisse, les agriculteurs jouent un rôle clé dans la réalisation des objectifs de conservation de la biodiversité. Mais comment faire pour qu'ils s'approprient ces objectifs? Dans une étude sociologique conjointe, l'Institut de recherche de l'agriculture biologique (FiBL) et la Station ornithologique suisse ont analysé l'influence d'une vulgarisation ciblée en promotion de la biodiversité sur l'attitude et les motivations des agriculteurs suisses. Les résultats fournissent de précieuses indications pour le travail de persuasion des vulgarisateurs en promotion de la biodiversité.

Introduction

Environ un tiers de la surface de la Suisse est utilisé à des fins agricoles et exploité de manière toujours plus intensive. Cette intensification se manifeste notamment par l'utilisation de gros engins puissants, une charge en bétail élevée, l'importation d'aliments concentrés, des apports en substances nutritives importants, mais aussi par la disparition d'éléments naturels du paysage. Cette évolution contribue à l'appauvrissement de la diversité des espèces dans les zones agricoles. Pour contrer cette perte, les agriculteurs doivent depuis 1993 exploiter au moins 7 % de leur surface agricole comme une surface de promotion de la biodiversité (SPB). L'application de cette mesure de compensation leur donne droit à des paiements directs. Toutefois, malgré le développement constant de ces paiements écologiques, la biodiversité en Suisse continue de décliner. Une mesure étatique possible serait de durcir la législation, par exemple en prescrivant un pourcentage de SPB plus élevé pour les surfaces agricoles ou en imposant une distance maximale entre deux SPB. Mais une telle proposition rencontrerait une forte opposition de la part des agriculteurs, qui tiennent à leur autonomie. Dès lors, des approches consistant à convaincre les agriculteurs de l'importance des mesures de promotion de la biodiversité et de leurs avantages économiques pour la production et l'exploitation paraissent plus prometteuses. Autrement dit, il faut que la protection de la biodiversité gagne leur cœur et leur raison. Cependant, on ne connaît pas encore le meilleur moyen pour pousser les agriculteurs à accorder plus d'estime à la biodiversité.

Cette étude visait trois objectifs. Premièrement, analyser si l'attitude et les motivations des agriculteurs étaient en rapport avec la proportion de SPB. Deuxièmement, évaluer si l'attitude et les motizu untersuchen ob Einstellung und Motivationen der Landwirt/innen im Zusammenhang mit dem Anteil der BFF stehen. Zweitens wurde evaluiert, ob Einstellung und Motivationen durch Beratung auf den Betrieben beeinflusst werden können. Das dritte Ziel war, herauszufinden, ob es Unterschiede zwischen Labelbetrieben (Bio, IP-Suisse) und Nicht-Label-Betrieben bezüglich Einstellung und Motivation gibt.

Stichprobe

Zwischen 2008 und 2015 wurde das Forschungsprojekt «Mit Vielfalt punkten - Landwirte und Landwirtinnen beleben die Natur» (MVP) mit 50 teilnehmenden Landwirt/innen durchgeführt. Von diesen wurden 25 zufällig ausgewählt und kontinuierlich im Bereich Biodiversität beraten. Diese 25 beratenen Landwirt/innen wurden angefragt, an einer sozialwissenschaftlichen Studie teilzunehmen. Davon stimmten 23 einer schriftlichen Befragung zu (Antwortquote 92%). Diese Stichprobe wurde ergänzt durch 110 Landwirt/innen (Antwortquote 36%), die entweder ohne Betriebsberatung am MVP-Projekt teilgenommen hatten oder nicht mit dem Projekt in Verbindung standen und zufällig ausgewählt wurden. In einer ersten Auswertung wurden keine Unterschiede in Bezug auf Einstellungen und Motivationen zwischen den MVP-Landwirt/innen ohne Beratung und den zufällig ausgewählten Landwirt/innen gefunden. Deswegen wurden diese zu einer Gruppe «nicht beraten» zusammengefasst. Alle Bio-und IP-Suisse-Landwirt/innen der Gesamtstichprobe (N=96) wurden der Gruppe «Labelbetriebe» zugeordnet. Die übrigen Landwirt/innen (N=37) bildeten die Gruppe «kein Label».

Resultate

Um die Einstellungen der Landwirt/innen zu prüfen wurden diese gefragt, wie stark sie 11 Aussagen zu den wichtigsten Aufgaben der Schweizer Landwirtschaft zustimmen. Die Ergebnisse sind in Abbildung 1 dargestellt. Aussagen, die mit dem Anteil an BFF signifikant korrelieren, wurden mit Sternen gekennzeichnet.Die Ergebnisse zeigen, dass einige Aussagen von allen Landwirt/innen ausdrücklich befürwortet wurden: unabhängig von der Produktionsweise oder ob sie eine gezielte Beratung erhalten hatten. Angesichts des hohen Masses an Zustimmung wäre es in diesem Fall für die Berater/innen wohl ohnehin schwierig, positive Effekte zu erzielen. Ein Beispiel dafür ist die Aussage: Eine wichtige Aufgabe ist die Erhaltung der Bodenfruchtbarkeit für die nächste Generation (Mittelwert 4,91), welche durch eine Beratung der Landwirt/innen wohl kaum noch mehr Zustimmung erhalten könnte. Eine Schlussfolgerung ist deshalb, dass die Aussagen, denen fast ausnahmslos voll zugestimmt wurde, einen Teil der Identität als Landwirte oder Landwirtin abbilden. Die deutlichsten Einstellungsunterschiede zwischen den beratenen und den nicht beratenen Landwirt/innen wurden bei den Aussagen mit der geringeren allgemeinen Zustimmung gefunden. Zwei Aussagen hatten eine geringe allgemeine Zustimmung und zeigten deutliche Unterschiede zwischen den beratenen und den nicht beratenen Landwirt/innen. Diese waren: Ökosystemleistungen wurden verbessert durch das Anlegen von Biodiversitätsförderflächen und

vations pouvaient être influencées par la vulgarisation dans les exploitations. Troisièmement, découvrir s'il existe des différences d'attitude et de motivation entre les exploitants certifiés (Bio, IP-Suisse) et non certifiés.

Échantillon

Le projet de recherche «Les paysans marquent des points, la nature gagne en diversité» a été mené entre 2008 et 2015 avec plus de 50 agriculteurs. Parmi ces participants, 25 ont été choisis au hasard pour être régulièrement conseillés en matière de biodiversité. Ces 25 agriculteurs ont ensuite été invités à participer à une étude sociologique, 23 ont ainsi accepté de répondre à un questionnaire écrit (taux de réponse de 92%). Cet échantillon a ensuite été complété par un second de 110 agriculteurs (taux de réponse de 36%), dont certains avaient participé au projet de recherche mais n'avaient pas bénéficié de conseils et d'autres n'étaient pas liés au projet et ont été choisis au hasard. Une première évaluation des données a permis d'observer qu'il n'y avait pas de différences d'attitude et de motivation entre les agriculteurs participant au projet mais ne bénéficiant pas de conseils et les agriculteurs choisis au hasard. Pour cette raison, ces deux groupes ont été réunis sous la catégorie «non conseillés». Les agriculteurs certifiés Bio ou IP-Suisse de l'échantillon global (N=96) ont été regroupés sous la catégorie «exploitation certifiée». Les agriculteurs restants (N=37) formaient, eux, le groupe «non certifiés».

Résultats

Pour évaluer le rapport des agriculteurs à la biodiversité, il leur a été demandé de noter dans quelle mesure onze affirmations correspondaient aux missions principales de l'agriculture suisse selon eux. Les résultats sont présentés dans la figure 1. Les affirmations qui ont une forte corrélation avec la proportion de SPB sont suivies d'astérisques. Les résultats montrent que l'ensemble des agriculteurs adhèrent à certaines affirmations, indépendamment de leur méthode de production ou du fait d'avoir été suivis par un vulgarisateur. Au vu des valeurs élevées pour ces affirmations, il serait bien difficile pour les vulgarisateurs de faire mouche sur ces points. Par exemple, l'affirmation «une mission importante consiste à conserver la fertilité du sol pour la génération suivante» (valeur moyenne 4,91) n'aurait pu obtenir une valeur plus élevée quand bien même les agriculteurs auraient été conseillés. On peut en conclure que les affirmations qui ont fait la presqu'unanimité expriment une partie de l'identité d'agriculteur. Les différences d'attitudes entre les participants qui ont bénéficié de conseils et ceux qui n'en ont pas bénéficié portent sur les affirmations qui ont reçu des valeurs plus basses en général. Deux affirmations, qui ont été moins bien notées par l'ensemble des participants, montrent des disparités considérables entre les agriculteurs conseillés et non conseillés. Il s'agit d'«aménager des SPB pour renforcer les prestations écosystémiques» et de «préserver et favoriser la biodiversité». Comme ces deux affirmations sont corrélées à la proportion de SPB, elles représentent des angles d'attaque potentiellement efficaces pour le travail de persuasion des vulgarisateurs.

Pour évaluer les motivations des agriculteurs, il leur a été demandé de noter dans quelle mesure ils étaient d'accord avec

Abb. 1: Mittelwerte der verschiedenen Gruppen. Höhere Antwortwerte zeigen eine stärkere Zustimmung zu der jeweiligen Aussage.

Was sind die wichtigen Aufgaben?

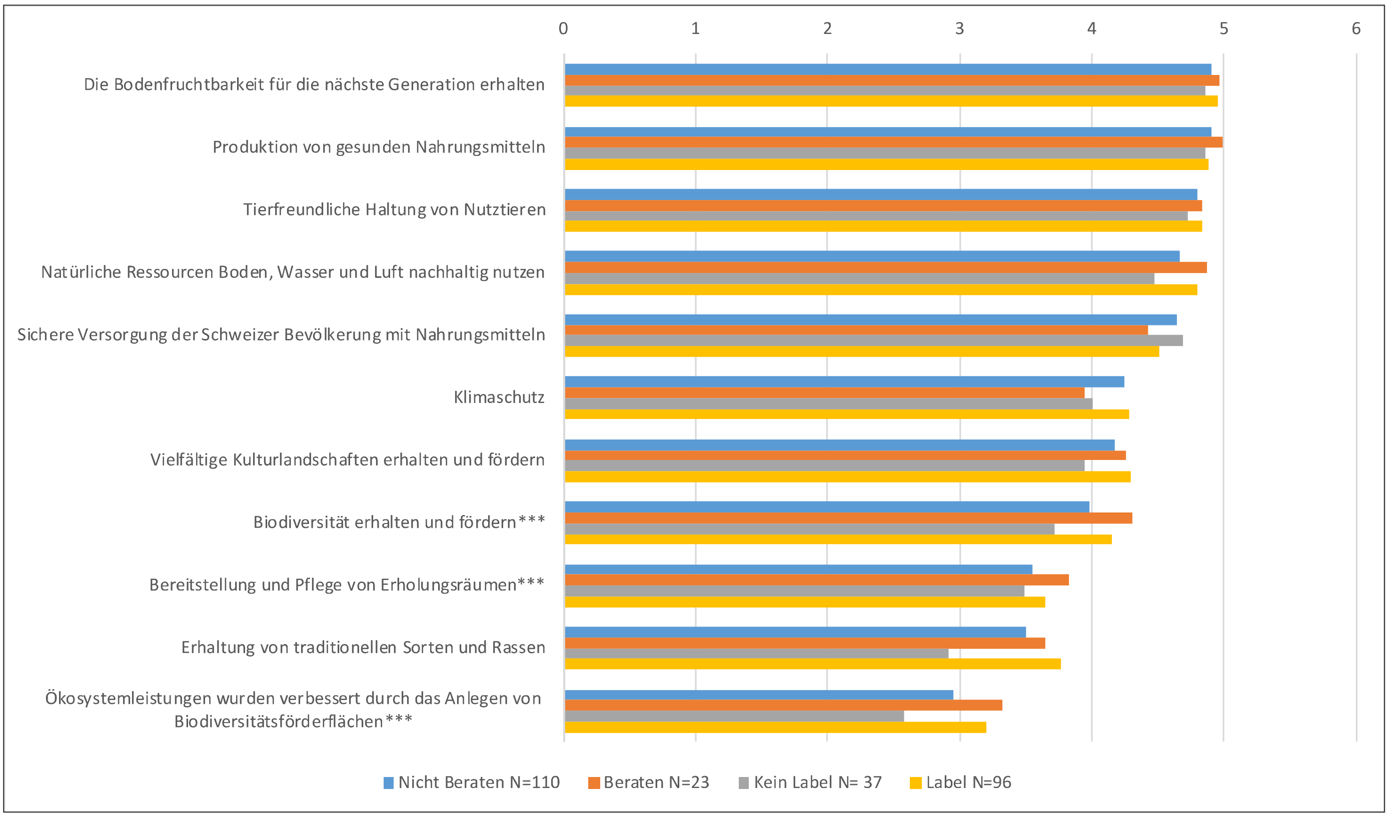
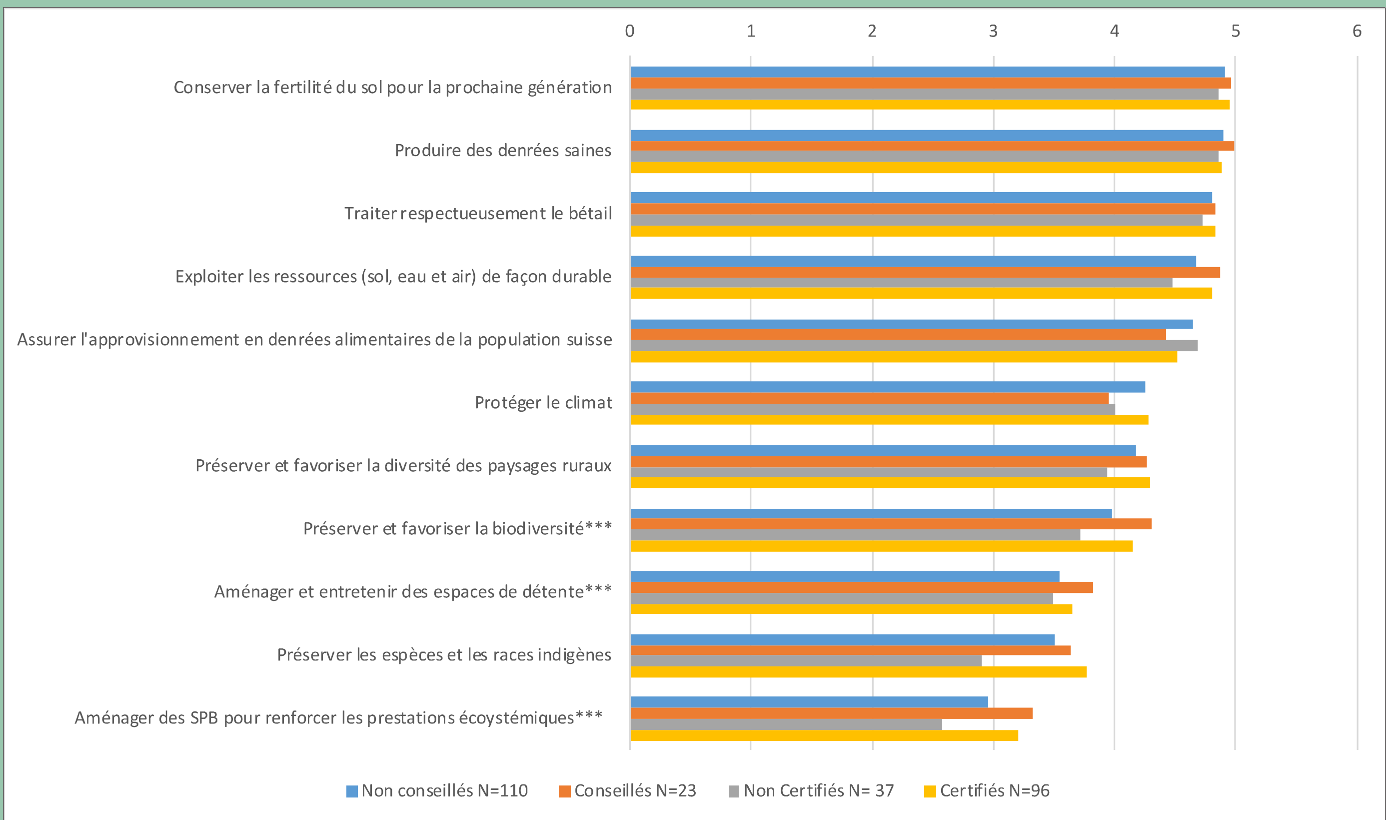


Fig.1: Valeur moyenne des différents groupes. Une valeur élevée indique que les agriculteurs adhèrent plus fortement à l'affirmation.

Quelles sont les missions principales de l'agriculture?



Biodiversität erhalten und fördern. Weil diese beiden Aussagen auch mit dem Anteil an BFF korrelieren bieten sie ein potentiell wirkungsvolles Ziel für die Überzeugungsarbeit von Berater/innen. Um die Motivationen der Landwirt/innen zu prüfen wurden diese gefragt, wie stark sie den 15 möglichen Gründen für das Anlegen von Biodiversitätsförderflächen zustimmen. Die Ergebnisse zur Motivation der Landwirt/innen sind in Abbildung 2 dargestellt. Aussagen, die mit dem Anteil an BFF signifikant korrelieren, wurden mit Sternen gekennzeichnet.

Verglichen mit den Fragen zur Einstellung waren bei den Motivationen generell eine geringere Zustimmung und grössere Unterschiede zwischen den Gruppen zu erkennen. Die beratenen Landwirt/innen zeigten im Allgemeinen eine höhere Zustimmung als die, die keine Beratung bekommen haben. Die Aussage, dass die Landwirtschaft eine ökologische Verantwortung trägt, ist mit der höchsten Zustimmung (Mittelwert 4,18) beantwortet worden. Bei den beratenen Landwirt/innen lag der Mittelwert gar bei 4,64. Die Antworten in Bezug auf die Motivationen korrelierten fast alle signifikant mit dem Anteil der BFF. Man kann daher schlussfolgern, dass die Beratung zu einer positiven Veränderung der Motivationen und schlussendlich zum Anlegen zusätzlicher BFF geführt hat.

Es ist aber theoretisch denkbar, dass sich hauptsächlich Landwirt/innen am Projekt beteiligt haben, die grundsätzlich eher positiv gegenüber den BFF eingestellt waren. Verglichen mit Kolleg/innen von Nicht-Label-Betrieben zeigten die Landwirt/innen von Labelbetrieben eine allgemein stärkere Zustimmung zu fast jeder Aussage, egal ob die Aussage zu Einstellungen oder Motivationen gehörte. Unsere Erklärung dafür ist, dass eine positive Einstellung zur Biodiversität zumindest einen Teil des Impulses für die Entscheidung, auf Bio- oder IP-Produktion umzusteigen, ausmacht. Zudem ist es möglicherweise eine Auswirkung der Beratung, die zur Umstellung auf Bio- oder IP-Produktion gehört. Es gibt aber Hinweise, dass diese Voreinstellungen nicht ausschliesslich bestimmend für die positiveren Einstellungen und Motivationen der Label-Landwirt/innen sind.

Zu Beginn des MVP-Projektes unterschied sich der Anteil BFF zwischen den später beratenen Landwirt/innen und denen, die keine Beratung bekommen haben, nicht. Zudem konnten keine Unterschiede in Bezug auf Einstellungen und Motivationen zwischen den nicht beratenen MVP- und den zufällig ausgewählten Landwirt/innen gefunden werden. Die beratenen MVP-Landwirt/innen steigerten dagegen im Verlauf der Untersuchung den Anteil und die Qualität der BFF markant. Auch war die Vielfalt der verschiedenen BFF-Typen deutlich grösser als bei Betrieben ohne Beratung. Bei diesen machen extensive Wiesen und Hochstammbäume den grössten Teil der BFF aus. Schliesslich lassen diese Ergebnisse den Schluss zu, dass die Einstellung und Motivation der Landwirt/innen massgeblich durch eine gezielte Beratung beeinflusst wurden und dies zu einem höheren Anteil an BFF sowie höherer Qualität auf den Betrieben geführt hat.

quinze raisons en faveur de l'aménagement de SPB. Les résultats quant aux motivations des agriculteurs sont présentés dans la figure 2. Les affirmations qui ont une forte corrélation avec la proportion de SPB sont suivies d'astérisques.

Par rapport au questionnaire sur l'attitude des agriculteurs, les questions sur leurs motivations présentaient en général des valeurs plus basses et de plus grandes différences entre les groupes. Les agriculteurs qui ont bénéficié de la vulgarisation adhéraient généralement plus aux affirmations que ceux qui n'en ont pas profité. L'affirmation selon laquelle l'agriculture a une responsabilité écologique est celle qui a reçu la plus haute valeur (valeur moyenne 4,18). Elle atteint même une valeur moyenne de 4,64 pour les agriculteurs qui ont été conseillés. Les affirmations en lien avec les motivations ont presque toutes une corrélation importante avec la proportion de SPB. On peut en conclure que la vulgarisation a conduit à une évolution positive des motivations et, in fine, à l'aménagement de SPB supplémentaires.

On peut théoriquement imaginer que ce sont principalement des agriculteurs fondamentalement favorables aux SPB qui ont participé à cette étude. Par rapport à leurs collègues non certifiés, les agriculteurs certifiés ont généralement adhéré plus fortement à presque toutes les affirmations, qu'elles portent sur l'attitude ou la motivation. Nous pensons à ce propos qu'une attitude positive envers la biodiversité motive, du moins partiellement, la décision de passer à une production biologique ou labelisée IP-Suisse. Il se peut aussi que la conversion à une production biologique ou labélisée IP-Suisse soit le résultat de la vulgarisation en promotion de la biodiversité. Certains signes montrent toutefois que la prédisposition ne détermine pas à elle seule l'attitude positive et les motivations des agriculteurs labélisés.

Au début du projet, la part de SPB entre les agriculteurs qui bénéficieront par la suite de la vulgarisation et ceux qui n'en bénéficieront pas ne présentait pas de différences. En outre, il n'a été observé aucune différence d'attitude et de motivation entre les participants au projet qui n'ont pas été conseillés et les agriculteurs qui ont été choisis au hasard. Par contre, les agriculteurs qui ont été conseillés ont augmenté la part et la qualité de leurs SPB durant la recherche de manière significative. La diversité de types de SPB s'est également révélée plus grande chez eux que chez les agriculteurs qui n'ont reçu aucun conseil, chez qui les prairies extensives et les peuplements d'arbres de haute tige constituaient la majorité des SPB. Finalement, ces résultats permettent de conclure que l'attitude et les motivations des agriculteurs ont été grandement influencées par une vulgarisation ciblée en promotion de la biodiversité et que cela a conduit à une augmentation de la part de SPB et de la qualité de celles-ci dans les exploitations.

Abb. 2: Mittelwerte der verschiedenen Gruppen. Höhere Antwortwerte zeigen eine stärkere Zustimmung zu der jeweiligen Aussage.

Ich lege freiwillig BFF an, weil...

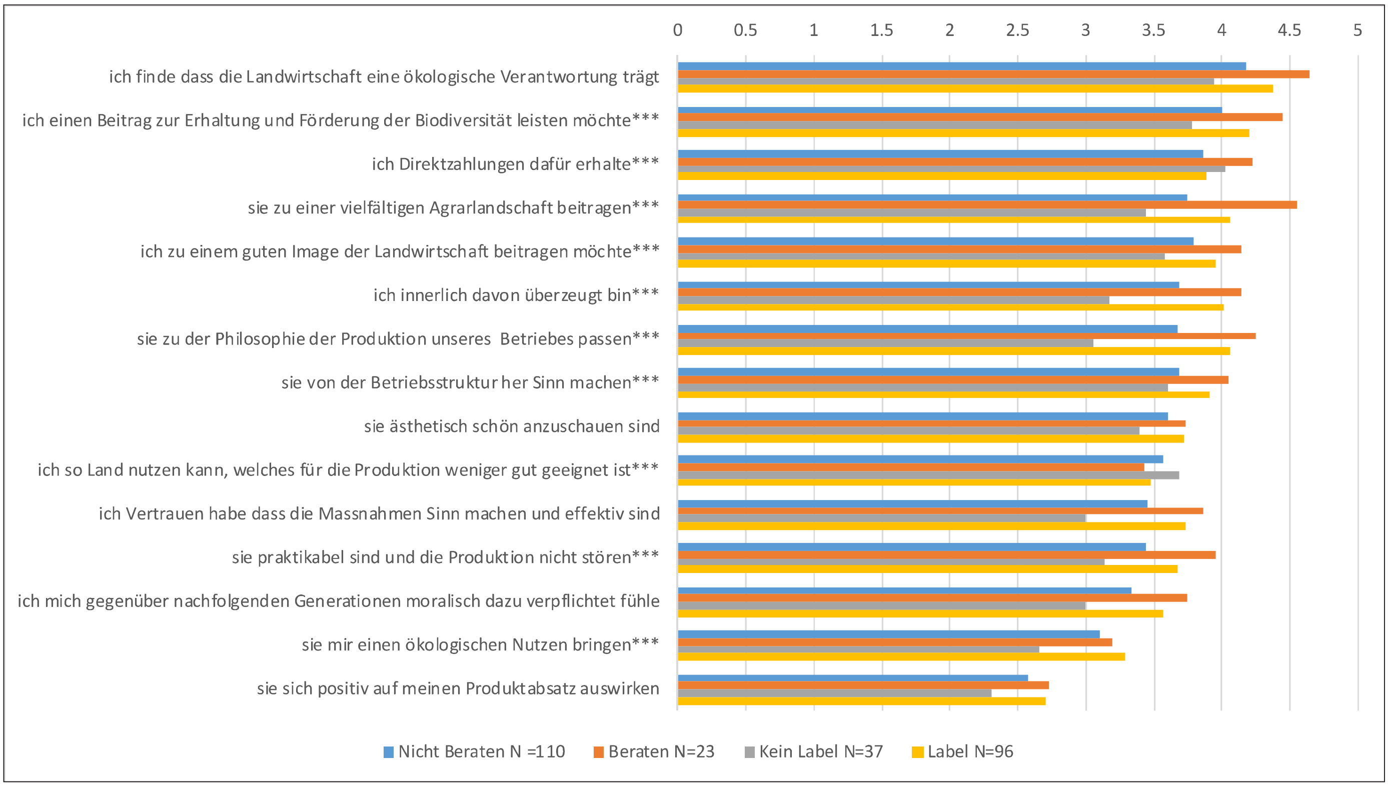
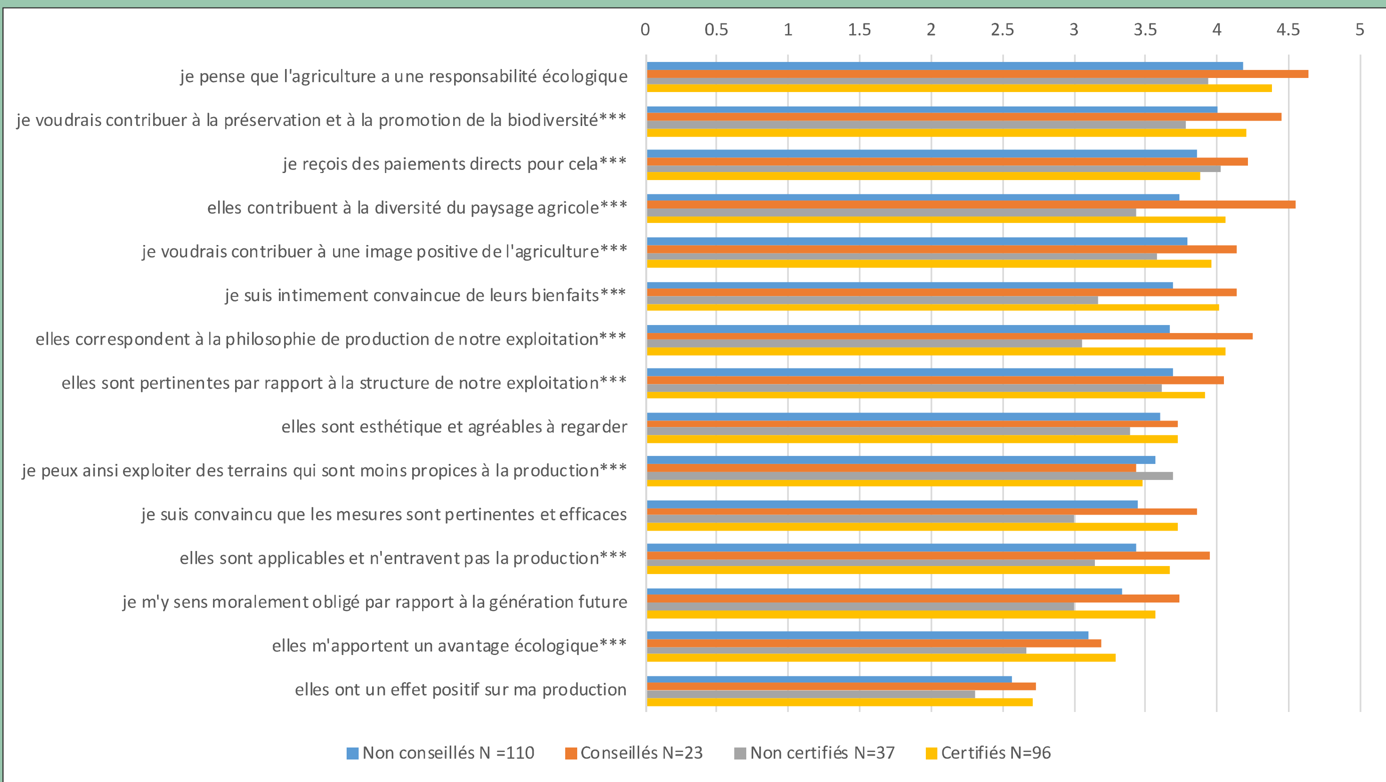


Fig. 2: Valeur moyenne des différents groupes. Une valeur élevée indique que les agriculteurs adhèrent plus fortement à l'affirmation.

J'aménage volontairement des SPB, car...



Eine ausführlichere Beschreibung der sozialwissenschaftlichen Studie finden sie im folgenden Artikel: (https://www.tandfonline.com/doi/abs/10.1080/1389224X.2018.1428205). Mehr Informationen über das MVP-Projekt und Links zu Publikationen finden Sie auf: http://www.fibl.org/de/schweiz/forschung/nutzpflanzenwissenschaften/pb-projekte/mvp.html

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Retrouvez une description plus détaillée de cette étude sociologique dans l'article disponible sous: https://www.tandfonline.com/doi/abs/10.1080/1389224X.2018.1428205 (en anglais). Pour plus d'informations sur le projet «Les paysans marquent des points, la nature gagne en diversité» et pour obtenir le lien de publications, rendez-vous sur: http://www.fibl.org/de/schweiz/forschung/nutzpflanzenwissenschaften/pb-projekte/mvp.html (en allemand)

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Anhang A: Fachliches Hintergrundwissen

Im Folgenden wird durch die Erläuterung von verwendeten Begrifflichkeiten und die Darstellung bestehender Zusammenhänge ein kontextbezogenes Hintergrundwissen vermittelt, das zum weiteren Verständnis der Arbeit beitragen soll. Dazu wird die aktuelle Forschungsliteratur genutzt.

Was ist Biodiversität?

Im Biodiversitätsübereinkommen der UN (*Convention on Biological Diversity*, CBD) wird Biodiversität wie folgt definiert: „Variabilität unter lebenden Organismen jeglicher Herkunft, darunter Land-, Meeres- und sonstige aquatische Ökosysteme und die ökologischen Komplexe, zu denen sie gehören. Dies umfasst die Vielfalt innerhalb der Arten (genetische Vielfalt) und zwischen den Arten (Artenvielfalt) und die Vielfalt der Ökosysteme (und entsprechend der Interaktionen darin)“ (United Nations, 1992). Zusätzlich wird in anderen Definitionen noch die „funktionelle Biodiversität“ eingeschlossen (Noss, 1990; Lyashevska und Farnsworth, 2012). Unter funktioneller Diversität werden der Wert, das Ausmaß und die Abundanz funktioneller Merkmale von Organismen in einem bestimmten Ökosystem zusammengefasst (TEEB, 2010). Diese Form der Biodiversität ist für die Landwirtschaft von großer Bedeutung, weil die dadurch generierten sogenannten „Ökosystemleistungen“ einen fundamentalen Baustein des Landbaus darstellen.

Daily (1997) umschreibt den Begriff Ökosystemleistung mit der Formulierung „Bedeutung von Biodiversität für den Menschen“. Etwas konkreter werden Cardinale *et al.* (2012), die Ökosystemleistungen als die durch ein Ökosystem generierten Vorteile für den Menschen definieren. Es können verschiedene Typen von Ökosystemleistungen unterschieden werden, nämlich „unterstützende“, „bereitstellende“, „regulierende“ und „kulturelle“ Leistungen (MA, 2005). Als Beispiele für unterstützende Ökosystemleistungen können Primärproduktion oder Nährstoffkreisläufe aufgeführt werden. Die Produktion von Fasern oder Nahrungsmitteln kann man hingegen den bereitstellenden Leistungen zuordnen. Ein Beispiel für eine regulierende Leistung ist die Schädlingsregulation. Die Erholungsfunktion für den Menschen ist eine kulturelle Ökosystemleistung (MA, 2005).

Eine der bekanntesten und eindrücklichsten Ökosystemleistungen mit immensem Einfluss auf die landwirtschaftliche Produktion ist die Bestäubungsleistung durch Insekten. Etwa 75 % der Kulturpflanzen weltweit sind auf eine Bestäubung durch Insekten angewiesen (TEEB, 2010).

Biodiversität umfasst also zahlreiche Aspekte und wird durch Klima, Geographie, Landschaft und Landnutzungsintensität beeinflusst. In Agrarökosystemen wird zudem häufig noch zwischen der „geplanten“ und der “assoziierten“ Biodiversität unterschieden (Altieri, 1999). Dabei wird unter „geplant“ diejenige Biodiversität verstanden, die unter dem direkten Management-Einfluss des Landwirtes steht (z. B. Kulturen und Tierbestand) und unter „assoziierter“ Biodiversität, die aus Interaktion mit den Kulturen und Nutztieren resultierende Einwanderung und Besiedlung der landwirtschaftlichen Nutzflächen aus der Umgebung, den Begleitbiotopen abseits der Produktionsbiotope (Altieri, 1999; Swift *et al.*, 2004; Tscharntke *et al.*, 2012).

Warum muss Biodiversität erhalten werden?

Jax (2002) weist darauf hin, dass die Suche nach dem Wert der Biodiversität sich im Grenzbereich zwischen Ethik und Ökologie befindet und dabei verschiedene Werte verbunden werden. Dazu gehören ökonomische, ästhetische und sich auf die menschliche Gesundheit und das Wohlbefinden beziehende Werte, aber auch solche, die sich nach dem Wohlbefinden und der Existenz anderer Lebewesen richten (Jax, 2002). Es ergeben sich somit zahlreiche mit diesen Werten verbundene Gründe, Biodiversität zu erhalten. So beschreiben Cardinale *et al.* (2012) beispielsweise „die Existenz des Lebens als die einzigartige Eigenschaft der Erde und als das Außergewöhnlichste am Leben, seine Vielfalt“.

In einem Artikel zur ethischen Bewertung der Biodiversität werden die folgenden Werte unterschieden (Hummel *et al.*, 1999):

- Lebenserhaltungswert
- Versicherungswert
- Ökonomische Werte
- Sozialer Annehmlichkeitswert
- Erholungswert

- Wissenschaftlicher Wert
- Ästhetischer Wert
- Kultureller Symbolisierungswert
- Biophiler Wert
- Transformativer Wert
- Religiöse oder spirituelle Werte

Ein Biodiversitätsverlust und der damit einhergehende Rückgang der Ökosystemleistungen können laut MA (2005) für den Menschen zu Gesundheitsbeeinträchtigungen, erhöhter Ernährungsunsicherheit, steigender Vulnerabilität, geringerer Materialverfügbarkeit, einer Verschlechterung der sozialen Beziehungen und einer geringeren Auswahl- und Aktionsfreiheit führen.

Welchen Einfluss hat die Landwirtschaft auf Biodiversität?

Die Wechselwirkungen zwischen Biodiversität und Landwirtschaft sind vielfältig. Die Abhängigkeit der Landwirtschaft von den durch Biodiversität generierten Ökosystemleistungen wurde bereits angesprochen. Die Landwirtschaft hat sowohl positive als auch negative Auswirkungen auf Biodiversität (Tscharntke *et al.*, 2005). So werden etwa 70 % des weltweiten Biodiversitätsverlustes auf eine intensivierte landwirtschaftliche Nutzung zurückgeführt (*Secretariat of the Convention on Biological Diversity* 2014). Die Landwirtschaft wird somit als der Haupttreiber des Biodiversitätsverlustes identifiziert (Butler *et al.*, 2007; Haines-Young, 2009; Bianchi *et al.*, 2013). Verschiedene Intensivierungsmerkmale wie gestiegene Stickstoff- und Pestizideinsätze tragen zu dieser Entwicklung bei. Für Benton *et al.* (2003) ist vor allem der Verlust von Landschaftsheterogenität zentral, der durch eine Entwicklung zu immer größeren Schlägen und einer Spezialisierung der Landnutzung mit einer Konzentration auf wenige Ackerkulturen entsteht. Dies führt zu Habitatverlusten in der Kulturlandschaft. Dennoch kann die Landwirtschaft grundsätzlich auch positive Effekte auf Biodiversität haben. Tscharntke *et al.* (2005) kritisieren, dass im Naturschutz der Fokus häufig auf die unberührte Natur gelegt wird, obwohl in landwirtschaftlich genutzten Gebieten ein Großteil der weltweiten Biodiversität zu finden ist und eine weniger intensive Landwirtschaft zu einer Steigerung der Habitatvielfalt und der Ökosystemfunktionen beitragen kann.

Produktionssysteme und Biodiversität

Es kann davon ausgegangen werden, dass je nach Intensität des Produktionssystems die Biodiversitätswirkung variiert. Dem Grundsatz des ökologischen Landbaus entsprechend ließe sich demnach vermuten, dass dem Biodiversitätsschutz eine größere Rolle beigemessen wird als in der konventionellen Landwirtschaft. So haben beispielsweise Schader *et al.* (2008) festgestellt, dass biologisch wirtschaftende Landwirte biodiversitätsfördernden Maßnahmen gegenüber aufgeschlossener sind. Der Unterschied zwischen ökologischer und konventioneller Landwirtschaft hinsichtlich der Auswirkungen auf Biodiversität wurde häufig untersucht (Hole *et al.*, 2005; Batáry *et al.*, 2013; Gabriel *et al.*, 2013; Tuck *et al.*, 2014). Bengtsson *et al.* (2005) sowie Tuck *et al.* (2014) wiesen nach, dass Betriebe des ökologischen Landbaus durchschnittlich eine etwa 30 % höhere Artenvielfalt als konventionelle Betriebe aufweisen. Der Biodiversitätseffekt in verschiedenen landwirtschaftlichen Systemen oder Intensitätsniveaus unterliegt allerdings einer großen Variation (Billeter *et al.*, 2008; Kleijn *et al.*, 2009; Hawes *et al.*, 2010).

Betriebsentscheidungen und Biodiversität

Allein aus dem großen Flächenanteil landwirtschaftlicher Nutzung ergibt sich die Schlüsselrolle der Landwirte für die Erhaltung der Biodiversität (Lokhorst *et al.*, 2011; Stoeckli *et al.*, 2017). Landwirte können neben der Wahl der Produktionsweise in vielfältiger Weise Einfluss auf Biodiversität nehmen. Gleichwohl kommt es häufig zu Zweifeln an der Wirksamkeit der Biodiversitätsmaßnahmen (Home *et al.*, 2014; Stoeckli *et al.*, 2017). Um die Landwirte von der Wirksamkeit der Biodiversitätsfördermaßnahmen zu überzeugen, sind die direkte Wahrnehmung positiver Effekte, der Einsatz von Biodiversitätsbewertungsmethoden auf Betriebsebene sowie begleitende Beratung potentiell geeignete Faktoren.

Biodiversität und Landschaftseffekte

Bei der Beobachtung von Biodiversitätsverlusten, aber vor allem auch beim gezielten Entgegenwirken, spielt die räumliche Betrachtungsebene (Feld-, Betriebs-, Landschaftsebene) eine wichtige Rolle. So können sich die Intensivierungseffekte durch die Landwirtschaft auf der Feldebene oder auf der Landschaftsebene ganz unterschiedlich zeigen und unterschiedliche Maßnahmen erfordern. Schneiders *et al.* (2011) weisen darauf hin, dass eine große Artenvielfalt auf der Landschaftsebene eine wichtige Rolle bei der Bereitstellung von Ökosystemleistungen spielt. Dabei wird die Artenvielfalt auf der Landschaftsebene von Billeter *et al.* (2008) sowie Walz und Syrbe (2013) als Funktion der Landnutzungsintensität und der Landschaftsheterogenität beschrieben. So kann beispielsweise durch komplexe Strukturen in der Agrarlandschaft die lokale Diversität gefördert und damit möglicherweise sogar eine intensive Nutzung teilweise kompensiert werden (Tscharntke *et al.*, 2005).

Biodiversitätsbewertung

Es wird deutlich, dass die Bewertung der Biodiversität allein schon wegen der Komplexität und der Abstraktheit des Begriffs schwierig ist (Walz und Syrbe, 2013). Dennoch ist die treffgenaue Bewertung eine Voraussetzung, um die Wirksamkeit der Fördermaßnahmen zu überprüfen. So könnten möglicherweise auch Landwirte, die häufig an der Wirksamkeit der Maßnahmen zweifeln (Home *et al.*, 2014; Stoeckli *et al.*, 2017), überzeugt werden. Auch bei der Bewertung der Umweltwirkungen von Produktionssystemen oder Produkten steigt das Interesse daran, die Auswirkung auf Biodiversität integrieren zu können (Lindqvist *et al.*, 2016).

Probleme der Biodiversitätsbewertung

Eine Hauptschwierigkeit der Biodiversitätsbewertung besteht darin, dass Biodiversität nicht direkt gemessen werden kann und es auch keine einzelne Messgröße gibt, mit der eine ganzheitliche Erfassung möglich ist (Magurran und McGill, 2011). Als Annäherung werden deswegen Biodiversitätsindikatoren benötigt. Aufgrund der Komplexität und der durchaus unterschiedlichen Biodiversitätsaspekte ist eine Kombination von Indikatoren für eine Bewertung geeigneter als lediglich ein einzelner Indikator (Bolger, 2001; Büchs, 2003; Stoddard *et al.*, 2006; Nielsen *et al.*, 2007; Flynn *et al.*, 2009; van Strien *et al.*, 2009; Heink und Kowarik, 2010; Beck *et al.*, 2012; Mandelik *et al.*, 2012). Daraus resultieren die üblichen Schwierigkeiten indikatorbasierter Ansätze.

Dazu zählen z. B. die Auswahl einer geeigneten Biodiversitätsmetrik oder die Übertragbarkeit von einer Indikator-Umweltbeziehung auf eine andere (Beck *et al.*, 2012). Auch die bereits unter dem Unterpunkt Landschaftseffekte angesprochene Betrachtungsebene ist ein wichtiges Element in der Bewertung, das immer bedacht werden muss. Dieser Sachverhalt verlangt die Auswahl einer angemessenen Betrachtungsskala vor deren Hintergrund die Biodiversitätsbewertung stattfinden soll (Feld *et al.*, 2009; Batáry *et al.*, 2012; McMahon *et al.*, 2012). Dieser Punkt ist insofern wichtig, als das Auffinden und Verhalten verschiedener Taxa sehr skalenabhängig ist. (Gabriel *et al.*, 2010). Eine weitere Schwierigkeit besteht in der Auswahl einer geeigneten Referenzsituation, die als Basiszustand herangezogen werden kann, um eine potentielle Biodiversitätsveränderung aufzeigen zu können (Nielsen *et al.*, 2007). Letztendlich ist für die Auswahl einer geeigneten Referenz auch die Fragestellung mitentscheidend, die mit Hilfe der Biodiversitätsbewertung beantwortet werden soll. So kann z. B. ein Zustand in der Vergangenheit (vor einer Landnutzungsänderung), ein natürlicher bzw. naturnaher Zustand oder ein angestrebter Zielzustand gewählt werden (Stoddard *et al.*, 2006). Eine andere Referenz kann das durchschnittliche Landnutzungsmuster einer Region sein. Es gibt keinen Referenzzustand, der sich für die Beantwortung aller möglichen Fragestellungen eignet.

Welche Biodiversitätsbewertungsmethoden gibt es?

Es gibt eine Reihe von Methoden, um die Biodiversitätswirkungen der Landwirtschaft zu bewerten (z. B. BIOBIO [Jeanneret *et al.*, 2012]; SALCA-Biodiversität [Jeanneret *et al.*, 2014], MANUELA [von Haaren *et al.*, 2012] etc.). Der vorgesehene Verwendungszweck der jeweiligen Methoden kann sich jedoch stark unterscheiden. So gibt es beispielsweise Methoden, die mit dem Ziel entwickelt wurden, Landwirte oder die Politik zu beraten, damit sich dadurch die Biodiversitätsleistungen verbessern. Andere Methoden wurden entwickelt, um spezifische Forschungsfragen beantworten zu können. Des Weiteren gibt es auch Biodiversitätsbewertungsmethoden, die einem reinen Monitoring dienen sollen oder solche, die eine Selbstbewertung der Biodiversitätsleistung auf dem eigenen Betrieb ermöglichen. Abgesehen von dem Verwendungszweck können sich die verschiedenen Methoden aber auch in anderen Charakteristiken unterscheiden, z. B. in der vorgesehenen Verwendungsebene (Produkt, Feld, Betrieb, Landschaft etc.), den betrachteten Biodiversitätsaspekten und

Indikatoren oder dem geographischen Rahmen, in dem die Biodiversitätsbewertungsmethoden angewandt werden können.

Besondere Aufmerksamkeit wird in dieser Arbeit der Biodiversitätsbewertung als mögliche Kategorie in Ökobilanzen zuteil. Eine Übersicht über die Entwicklungen innerhalb dieser Umweltbewertungsmethode und speziell vor dem Hintergrund einer Bewertung von landwirtschaftlichen Produkten wird in Publikation III, „The challenges of including impacts on biodiversity in agricultural life cycle assessments“, gegeben. Im folgenden Abschnitt wird die Ökobilanzmethode daher kurz näher erläutert.

Was sind Ökobilanzen?

Die Ökobilanz ist eine Methode, mit der die Umweltwirkungen eines Produktes oder Produktionssystems abgeschätzt werden können. Häufig wird auch der Begriff Lebenszyklusanalyse (*Life Cycle Assessment,* LCA) verwendet, da in der Regel der gesamte „Lebenszyklus“ von der Produktion über die Nutzungszeit bis hin zur Entsorgung eines Produktes betrachtet wird. Die Ökobilanz kann verwendet werden, um die Ökoeffizienz verschiedener Prozesse und Produkte zu ermitteln. Dazu werden die Umweltauswirkungen während der Herstellung eines Produktes in Beziehung zum Produktionswert gesetzt. Die Ökobilanz eignet sich aber auch, um verschiedene Produktionsweisen miteinander zu vergleichen. Die Umweltwirkungen werden dabei auf eine funktionelle Einheit bezogen (z. B. kg/Produkt oder Flächeneinheit).

Die Grundsätze und die Rahmenbedingungen von Ökobilanzen sind in der Norm ISO 14040 (*International Standard Organisation*, 2006a) beschrieben und definiert, die Anforderungen und Richtlinien in der Norm ISO 14044 (*International Standard Organisation,* 2006b). In Ökobilanzen werden vier verschiedene Phasen unterschieden: die Definition von Ziel und Umfang, die Bestandsanalyse, die Folgenabschätzung und die Interpretationsphase

Ökobilanzierung landwirtschaftlicher Produkte

Die Anwendung der Ökobilanz zur Beurteilung landwirtschaftlicher Produkte und Prozesse und damit auch die Möglichkeit eines Vergleiches verschieden produzierter landwirtschaftlicher Erzeugnisse hinsichtlich ihrer ökologischen Auswirkungen hat in den letzten Jahren stetig zugenommen (Meier *et al.*, 2015; Foteinis und Chatzisymeon, 2016; Tricase *et al.*, 2017). Häufig werden jedoch bei der Ökobilanzierung im Zusammenhang mit der Landwirtschaft wichtige Umweltwirkungen wie Biodiversität, Bodenqualität und Tierwohl nicht standardmäßig abgebildet (Geier, 2000; Hayashi *et al.*, 2006). Dieser Umstand führt dazu, dass das durch die Ökobilanzen widergespiegelte Bild der Landwirtschaft im Hinblick auf wesentliche von ihr verursachte Umweltwirkungen unvollständig ist, woraus Probleme für Vergleiche zwischen landwirtschaftlichen Systemen und falsche Schlussfolgerungen resultieren. Von wissenschaftlicher Seite wird deshalb seit langem eine methodische Weiterentwicklung hinsichtlich der Wirkungsabschätzung spezifischer landwirtschaftlicher Aspekte vorgeschlagen (Haas, 2003; Milà i Canals *et al.*, 2007). Aktuell gibt es dazu einige Forschungstätigkeiten, die sich dem Thema Biodiversität und *Life Cycle Assessment* (LCA) widmen (Curran *et al.*, 2016; Teixeira *et al.*, 2016; Lüscher *et al.*, 2017; Trydeman Knudsen *et al.*, 2017).

Anhang B: Methodisches Hintergrundwissen

In diesem Abschnitt wird Hintergrundwissen zu den wissenschaftlichen Methoden geliefert, die in den verschiedenen Publikationen verwendet wurden. Dabei werden die Gründe für die Auswahl und die Charakteristiken der Methoden beschrieben, die im Gesamtprojekt dieser Dissertation eingesetzt wurden.

<u>Quantitative Methoden</u>

Die Wahl von quantitativen Methoden für die ersten beiden Publikationen (I, II) ergibt sich daraus, dass diesen bereits eine qualitative Studie (Home *et al.*, 2014) vorausgegangen war, in der Hypothesen aufgestellt wurden, die nun mit Hilfe einer quantitativen Studie geprüft werden sollten.

Es ist bekannt, dass sich quantitative Studien mit Hilfe der Inferenzstatistik dazu eignen, bereits aufgestellte Hypothesen zu überprüfen, Zusammenhänge (Korrelationen) zu identifizieren und nachzuweisen sowie möglicherweise Verallgemeinerungen zu treffen (Creswell und Creswell, 2017). Ein angemessen großer Stichprobenumfang gehört dabei zu den Voraussetzungen (Patten und Newhart, 2017).

Statistische Methoden zur Datenanalyse

Für die statistische Auswertung in den Publikationen I und II wurden nicht-parametrische Tests ausgewählt, mit der Begründung, dass bei der Verwendung einer fünfstufigen Likert-Skala, die in den Fragebögen durchgehend als Antwortmöglichkeiten zur Auswahl stand, nicht von einer Normalverteilung ausgegangen werden konnte. Genau diese Frage, ob es sich bei der Verwendung einer Likert-Skala um eine Normalverteilung handelt oder nicht, hat jedoch unter Sozialwissenschaftlern im Laufe der Jahre für erhebliche Diskussionen gesorgt. Allerdings haben einige Statistiker darauf hingewiesen (Sullivan und Artino, 2013), dass auch parametrische Tests (bei denen eine Normalverteilung vorausgesetzt wird) funktionieren und dass die Unterschiede in den Ergebnissen in der Regel vernachlässigbar sind – vor allem, wenn die Stichprobe ausreichend groß ist. Trotzdem wurden für diese Arbeit nicht-parametrische Tests gewählt. Für den Gruppenvergleich wurde der Mann-Whitney-U-Test und für die Korrelationen die Rangkorrelationsanalyse nach Spearman angewandt. Alle statistischen Tests wurden mit dem Programm SPSS Version 17 durchgeführt.

Mann-Whitney-U-Test (Publikation I, II)

Der Mann-Whitney-U-Test ist das nicht-parametrische Äquivalent zum T-Test und dient dazu, zwei unabhängige Stichproben zu vergleichen. Er wird angewandt, wenn die Voraussetzungen (Normalverteilung) für einen T-Test nicht gegeben sind. Die Voraussetzung für die Verwendung des Mann-Whitney-U-Tests ist, dass die abhängige Variable ordinalskaliert ist. Diese Voraussetzung ist durch die genutzte Likert-Skala erfüllt.

Rangkorrelationsanalyse nach Spearman (Publikation I)

Bei der Rangkorrelationsanalyse nach Spearman wird das Maß ermittelt, das den Zusammenhang zwischen zwei Variablen beschreibt (Stahel, 2013). Genau wie beim Mann-Whitney-U-Test handelt es sich um einen nicht-parametrischen Test, bei dem die gleichen Voraussetzungen erfüllt werden müssen. Die parametrische Entsprechung ist hier die Bravais-Pearson-Korrelationsanalyse. Der einzige Unterschied zum Bravais-Pearson-Test ist, dass nicht mit den originalen Messdaten, sondern mit deren Rängen gerechnet wird. In unseren Analysen wurden mögliche Korrelationen zwischen Motivations- und Einstellungsparametern und dem Flächenanteil mit Biodiversitätsförderung auf den jeweiligen Betrieben ermittelt. Der Wert des Korrelationskoeffizienten liegt zwischen 1 und -1. Je mehr er sich 0 nähert, desto schwächer ist die Beziehung zwischen den analysierten Variablen.

Hauptkomponentenanalyse (Publikation I)

Ziel der Hauptkomponentenanalyse war es, zu überprüfen, ob die zahlreichen Motivationsfaktoren in wenige Hauptkomponenten zusammengefasst werden können. Dabei wird ein mathematisches Verfahren genutzt, um zu prüfen, ob verschiedene, möglicherweise korrelierte Variablen in eine kleinere Anzahl von nicht korrelierten Variablen zusammengefasst werden können, um damit Erklärungen für mögliche Muster in den Datensätzen ableiten zu können (Hotelling, 1933). Diese kleinere Anzahl von repräsentativen Variablen muss aber die maximal mögliche Varianz im ursprünglichen Datensatz berücksichtigen (James *et al.*, 2013). Der kumulierte Prozentsatz der erklärten Varianz gibt einen Hinweis auf die Passgenauigkeit der Komponentenlösung (Lorenzo-Seva, 2013). Varimax ist ein Algorithmus, der versucht, die Hauptkomponenten so zu drehen, dass einzelne Variablen tendenziell nur einer Komponente zugeordnet werden (Fabrigar *et al.*, 1999). Die Faktorladungen, in der Hauptkomponentenanalyse auch Komponentenladungen genannt, sind die Korrelationskoeffizienten zwischen den Variablen und dem Faktor (Komponente) (Fabrigar *et al.*, 1999), die darstellen, wie umfänglich ein Faktor eine Variable in der Hauptkomponentenanalyse erklärt.

Je höher eine Faktorladung ist, desto größer ist der Zusammenhang zwischen der Variable und dem Faktor. Die Werte liegen zwischen -1 und 1. Je näher sie an 0 liegen, desto geringer ist der Zusammenhang. In Fällen, in denen zwei Faktorenladungen für eine Variable von ähnlicher Größe sind, kann die Variable als von beiden Komponenten gleichermaßen beeinflusst angesehen werden (Abdi und Williams, 2010).

Als Voraussetzung zur Durchführung der Hauptkomponentenanalyse wurden die folgenden Untersuchungen vorangestellt:

Cronbachs Alpha

Cronbachs Alpha gibt das Maß der internen Konsistenz einer Skala an und wird verwendet, um die Zuverlässigkeit unserer Motivationsskala zu unterstützen. Theoretisch können Werte zwischen negativ unendlich und 1 erreicht werden. In der Literatur werden Schwellenwerte ab 0,7 genannt, die aber gleichzeitig auch stark diskutiert werden (Schecker, 2014).

Kaiser-Meyer-Olkin

Mit dem Kaiser-Meyer-Olkin-Test (KMO) wurde geprüft, ob sich der Datensatz für eine Faktorenanalyse eignet. Der KMO variiert zwischen 0 und 1 und sollte als Voraussetzung mindestens 0,60 betragen, um eine Faktorenanalyse durchzuführen (Tavakol und Dennick, 2011).

Bartlett-Test

Der Bartlett-Test ist ein weiterer Test, der ebenfalls angewandt wird, um zu prüfen, ob sich ein Datensatz für eine Hauptkomponentenanalyse eignet (Fabrigar *et al.*, 1999). Mit diesem Test wird geprüft, ob sich die Korrelationskoeffizienten der Korrelationsmatrix von 0 unterscheiden. Eine Faktorenanalyse ist nämlich nur dann sinnvoll, wenn zwischen den untersuchten Variablen und mindestens einigen anderen untersuchten Variablen überhaupt Korrelationen zu finden sind (Janssen und Laatz, 2009). Das Ergebnis ist ein Chi-Quadrat-Wert und der Test sollte signifikant sein ($p<0,05$) (Williams *et al.*, 2010).

Eigenwert

Der Eigenwert (Eigenvalue) entspricht der Summe der quadrierten Faktorenladungen einer Komponente (Abdi und Williams, 2010) und gibt an, wie groß der Anteil der Gesamtvarianz aller Variablen ist, der durch die Komponente erklärt wird. Das „Kaiser-Kriterium“, welches auch Eigenwert-Regel genannt wird, gibt vor, dass nur Faktoren beibehalten werden sollen, die einen Eigenwert >1,0 aufweisen (Costello und Osborne, 2005).

Theorie des überlegten Handelns (Publikation I)

Die „Theorie des überlegten Handelns“ (Ajzen, 1991) bietet einen methodischen Rahmen, damit die Daten eingeordnet und interpretiert werden können. Die Theorie wurde auch schon häufiger eingesetzt, um bei verschiedenen Fragestellungen die Entscheidungsfindung von Landwirten zu analysieren (Zubair und Garforth, 2006; Schroeder *et al.*, 2015; van Dijk *et al.*, 2016). Bei dieser Theorie wird davon ausgegangen, dass die Absicht, sich auf etwas einzulassen, auf drei Faktoren basiert. Diese sind die Einstellung gegenüber dem Verhalten, die sozialen Normen und die wahrgenommene Verhaltenskontrolle. Zu einem späteren Zeitpunkt wurde die Theorie noch um die tatsächliche Verhaltenskontrolle ergänzt (Ajzen, 2002). Im Rahmen unserer Fragestellung wurde in Publikation I anhand dieser Theorie überprüft, ob es sich bei „Direktzahlungen“ wie bei vielen anderen Motivationsfaktoren um eine wahrgenommene Verhaltenskontrolle handelt oder ob diese gesondert betrachtet werden sollten.

Literaturübersicht (Publikation III)

In der Publikation „The challenges of including impacts on biodiversity in agricultural life cycle assessments“ wurde die Methode einer systematischen Literaturübersicht angewandt, um die aktuellen Entwicklungen bzw. Methoden und Methodenentwicklungen zur Biodiversitätsbewertung innerhalb von Ökobilanzen zusammenzufassen.

Dazu wurden in der elektronischen Datenbank „ISI Web of Science“ die Suchbegriffe *„biodiversity“* (Biodiversität) *und „life cycle assessment“* (Ökobilanz) eingegeben, um geeignete Studien zu finden. Des Weiteren wurden auch Studien miteinbezogen, die im Literaturverzeichnis der bereits durch die Datenbanksuche gefundenen Artikel vermerkt waren. Ziel der Literaturübersicht war es, einen Überblick über den aktuellen Stand der Methodenentwicklung zu geben und zu analysieren, inwieweit die Methoden geeignet sind, um die Biodiversitätswirkung landwirtschaftlicher Produkte zu bewerten. Neben Publikationen aus wissenschaftlichen Zeitschriften wurden auch Berichte und Konferenzbeiträge akzeptiert.

Fallstudie (Publikation IV)

In Publikation IV wurde eine Fallstudie als Methode gewählt. Nach Yin (2011) ist eine Fallstudie eine empirische Studie, die ein Phänomen in einem realen Kontext untersucht. Fallstudien werden insbesondere dann angewandt, wenn die Grenzen zwischen dem Kontext und dem Phänomen nicht ganz klar sind (Yin, 2011). Es ist nicht das Ziel einer Fallstudie, Generalisierungen abzuleiten (Hays, 2004). Ein wichtiges Ziel ist hingegen, Aufmerksamkeit zu generieren und Vorschläge für spezifische Zusammenhänge abzuleiten. In dieser Studie sollten die Unterschiede der verschiedenen Methodenansätze herausgearbeitet und die Ergebnisse miteinander sowie mit Biodiversitätserhebungen auf den Betrieben verglichen werden.

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