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LIFE CYCLE ASSESSMENT OF SWISS AGRICULTURE UNDER CLIMATE CHANGE AND THE IMPACTS OF WATER USE ON AQUATIC BIODIVERSITY

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ABSTRACT

Climate change is expected to affect agriculture worldwide in the coming century, provoking decreases in yields in many regions. In parallel, an increasing global world population will increase demand for food production. By 2050, Switzerland is expected to face dryer and hotter summers, and decreased yields, in the productive lowland regions. Switzerland has a political target of agricultural self-sufficiency, and therefore has an interest in maintaining yields; farmers also need to maintain their revenue at a profitable level. Swiss agriculture should therefore adapt to climate change, through management changes at the farm scale and new policies and incentives at the regional and national scale. To respect environmental targets, these adaptation strategies should also avoid increasing environmental impacts. This thesis used life cycle assessment to assess the environmental impacts of Swiss agricultural adaptation scenarios to climate change at the farm and regional scale for two regions in Switzerland. One important adaptation option identified is the increased use of river water for irrigation, which may affect aquatic biodiversity. As there was no existing satisfactory method to assess these impacts, we developed a new impact assessment method. The new method uses species-discharge relationships and is applicable over large spatial regions. In order to investigate a potential causal impact pathway, an alternative assessment approach was also developed using river water temperature modeling. We additionally assessed the use of groundwater and the riparian shading of the river as alternative or complimentary watershed management options.

Our main findings show that under the future climate scenarios, agricultural productivity for human consumption is expected to decrease if farmers are left to economically optimize their farms, and most environmental impacts relative to productivity will increase, resulting in a decrease in environmental efficiency. In order to maintain productivity at a regional scale, agriculture would make intensive use of water for irrigation. A major trade-off between yields and aquatic biodiversity is therefore found in Swiss agricultural adaptation to climate change. It is therefore recommended that aquatic biodiversity impacts be considered in assessments of agriculture under future climate scenarios. Potential species loss in one case study region reached 8% of species in the studied catchment, in contrast with the low water stress index currently attributed to the region. In general, other impacts per amount produced (such as global warming potential) decrease if future productivity is maximized, whereas strategies leading to a minimization of environmental impacts cause a large drop in productivity. In particular, policies directly targeting the restriction of water use for irrigation are very efficient at reducing impacts on aquatic biodiversity, however, they result in a decrease in productivity. Decreasing yields may lead to

increases in imports and environmental impacts induced at the sites of production. Outcomes are largely sensitive to policy choices as well as climate change, and there is therefore a high potential for policy-makers to influence and mitigate the effects of climate change on agricultural productivity and the associated environmental impacts. At the farm scale, outcomes vary according to region and farm type, and policies should be tailored to these differences. The balance between mitigated aquatic biodiversity impacts and agricultural productivity should involve multiple management options, including options beyond agricultural management (such as water resource management options, e.g. use of groundwater, riparian shading of rivers). Any single approach is not sufficient to avoid all impacts, and a mitigation strategy should involve a combination of options (other options include limitation of irrigation through crop choice and intensity management, channel restoration, provision of biodiversity shelters, and use of lake water for irrigation).

The assessment of impacts of river water withdrawals on aquatic biodiversity using species-discharge relationships is sensitive to the spatial resolution used: consideration of the location of withdrawals within a river basin (as proposed here) significantly affected the magnitude of impact. The inclusion of additional taxa is highly relevant if considering absolute rather than relative potential species loss. Absolute potential species loss, weighted with an indication of species vulnerability, was proposed here, and is recommended in order to provide a relationship with an absolute reference such as global species extinction. Further aspects that influence the magnitude of potential impacts are the use of region-specific species-discharge relationships and the choice of regression function used to model the relationship. The developed impact assessment method has a high potential for developing characterization factors across broad spatial extents. However, it is not recommended for local assessments and watershed management (since this may require more spatial and temporal detail in order to assess the benefits of specific mitigation measures), and cannot prove the existence of a causal relationship between species richness and in-stream discharge. The assessment of impacts of river water withdrawals on aquatic biodiversity using a deterministic river water temperature model, on the other hand, was found to be better adapted to local watershed management decision-making, and models a causal relationship. The level of detail required was however not considered adequate for broad spatial coverage (such as encountered in life cycle assessment), despite efforts to minimize computational and data requirements. Outcomes may be highly site-specific, with temperature changes in the case study river shown to be of little concern compared to changes in discharge per se.

We highly recommend the consideration of water use impacts in the life cycle assessment of agriculture in the context of climate change. In particular, the location of withdrawals within a basin and impacts on aquatic biodiversity are key parameters. Agricultural adaptation and watershed management should make use of complimentary solutions in order to ensure the fulfilling of multiple goals (such as economic profitability, food productivity and maintenance of environmental quality). Finally, extended consequences of regional scale decisions in a broader national and international context, as well as the social implications of adaptation strategies, should be investigated in future in order to provide a complete sustainability assessment of agricultural adaptation to climate change.

ZUSAMMENFASSUNG

Es ist zu erwarten, dass der Klimawandel im kommenden Jahrhundert die Landwirtschaft weltweit betreffen und zu Ertragsabnahmen in vielen Regionen führen wird. Gleichzeitig wird eine erhebliche Zunahme der Weltbevölkerung prognostiziert, was zu einem erhöhten Bedarf an Nahrungsmitteln führen wird um die Ernährungssicherung für alle zu garantieren. In der Schweiz ist davon auszugehen, dass trockenere und heissere Sommer bis 2050 in den hochproduktiven Flachlandgebieten zu niedrigeren Erträgen führen werden. Angesichts des Selbstversorgungsziels der Schweiz ist diese Entwicklung umso verheerender und eine Erhaltung der gegenwärtigen Erträge wäre von grossem Interesse. Gleichzeitig haben Landwirte Interesse daran, ihr Einkommen auf profitablem Niveau zu halten. Die Schweizerische Landwirtschaft sollte sich daher durch Anpassungen auf Ebene der landwirtschaftlichen Betriebsführung, sowie durch politische Strategien und Anreize auf regionaler und nationaler Ebene dem Klimawandel anpassen. Dabei ist darauf zu achten, dass diese Anpassungsstrategien nicht zu erhöhten Umweltauswirkungen führen und damit andere Umweltziele gefährden. Diese Dissertation hat die Umweltauswirkungen von Anpassungsstrategien der Schweizerischen Landwirtschaft auf den Klimawandel auf Betriebs- und regionaler Ebene anhand der Methode der Ökobilanzierung für zwei Fallstudien in verschiedenen Regionen der Schweiz analysiert. Eine wichtige identifizierte Möglichkeit zur Anpassung an den Klimawandel ist eine erhöhte Nutzung der Fliessgewässer für Bewässerungszwecke, wobei mit einem Einfluss auf die aquatische Biodiversität zu rechnen ist. Da eine befriedigende Methode zur Abschätzung solcher Auswirkungen zu Beginn dieser Dissertation nicht existierte, wurde basierend auf Arten-Abfluss Kurven eine Methode entwickelt, die räumlich breit anwendbar sein sollte. Parallel dazu wurde eine alternative Methode entwickelt, die einen kausalen Zusammenhang zwischen der modellierten Wassertemperatur und möglichen Umweltauswirkungen herstellt. Zusätzlich wurde auch die Nutzung von Grundwasser anstelle von Flusswasser und die Beschattung des Flusses als alternative oder komplementäre Wasserbewirtschaftungsoptionen analysiert.

Die Ergebnisse zeigen, dass die Produktion von Nahrungsmitteln unter zukünftigen klimatischen Bedingungen abnehmen wird, wenn Landwirte ihre Betriebe lediglich ökonomisch optimieren. Auf die Produktivität bezogen hingegen ist zu erwarten, dass die Umweltauswirkungen zunehmen werden und damit die Umwelteffizienz der Produktion insgesamt abnehmen wird. Um die Produktivität auf regionaler Ebene zu erhalten, müsste die Landwirtschaft verstärkt Wasser für Bewässerungszwecke einsetzen. Dies stellt einen massgeblichen Trade-off hinsichtlich der Anpassung der Schweizerischen

Landwirtschaft auf den Klimawandel dar, welcher sich in Form von Auswirkungen auf aquatische Biodiversität zeigen könnte. Daher sollten solche Auswirkungen in Nachhaltigkeitsanalysen von zukünftigen landwirtschaftlichen Systemen berücksichtigt werden. Der potenzielle Artenverlust betrug in einer Fallstudie 8% der vorkommenden aquatischen Arten innerhalb des betrachteten Einzugsgebietes und stand damit gleichzeitig im Gegensatz zum tiefen Wasserstressindex für dieses Gebiet. Mit Ausnahme dieser Auswirkungen des Wasserverbrauchs ist generell davon auszugehen, dass bei einer Maximierung der landwirtschaftlichen Produktivität, die Umweltauswirkungen (wie z.B. das Treibhausgaspotenzial) bezogen auf die produzierte Menge im zukünftigen Klima abnehmen werden (durch die Steigung der Produktion und der Effizienz), wohingegen Strategien zur Minimierung der Umweltauswirkungen eine erhebliche Verringerung in der Produktivität zur Folge haben könnten. Dies kann zu einer Zunahme der Nahrungsmittelimporte und folglich zu einer Verlagerung der Umweltauswirkungen am an den Herstellungsort führen. Da sowohl Politik- als auch Klimaszenarien einen starken Einfluss auf diese Ergebnisse haben, besteht ein grosses Potenzial durch entsprechende politische Massnahmen die Auswirkung des Klimawandels auf die landwirtschaftliche Produktivität und die Umwelt zu vermindern. Da auf Betriebsebene die Ergebnisse je nach Gebiet und Betriebstyp variieren, sollten folglich auch bei der Umsetzung politischer Massnahmen lokaler Faktoren berücksichtigt werden. Politische Massnahmen, die direkt auf die Begrenzung der Wasserentnahme abzielen, sind bezüglich der Verminderung der Auswirkungen auf die aquatische Biodiversität sehr effizient. Gleichzeitig resultieren sie jedoch auch in einer Abnahme der Produktivität und lösen den Trade-off zwischen Produktivität und Auswirkungen auf die aquatische Biodiversität nicht. Die simultane Verminderung der Auswirkungen auf die aquatische Biodiversität und die Erhaltung der Produktivität auf regionaler Ebene bedarf also mehrerer, weitergreifender Massnahmen, wie z.B. dem Einsetzen von Grundwasserressourcen und der Flussbeschattung. Ein einzelner Ansatz reicht nicht aus, um alle Umweltauswirkungen zu vermeiden: eine Strategie zur Anpassung an den Klimawandel sollte also eine Kombination mehrerer Massnahmen beinhalten (weitere Massnahmen sind z.B. die Reduktion der Bewässerung durch eine entsprechende Auswahl der Kulturen und Intensitätsmanagement, die Renaturierung von Flüssen, die Bereitstellung von Schutzzonen für Biodiversität und die Nutzung von Seewasser zur Bewässerung).

Die Abschätzung der Auswirkungen der Entnahme von Flusswasser auf die aquatische Biodiversität anhand von Arten-Abfluss Kurven ist ebenfalls abhängig von der räumlichen Auflösung: die Berücksichtigung des Entnahmeorts innerhalb eines Einzugsgebiets (wie in dieser Dissertation vorgeschlagen) hat das Ergebnis signifikant beeinflusst. Die Berücksichtigung zusätzlicher

Indikatorarten ist höchst relevant, wenn die absolute anstelle der relativen Anzahl verlorener Arten betrachtet wird. Die absolute Anzahl potenziell verlorener Arten, gewichtet mittels der Vulnerabilität der Arten, wurde in dieser Dissertation vorgeschlagen und empfiehlt sich, um eine Beziehung zu einer absoluten Referenz, wie dem globalen Artensterben, herzustellen. Weitere Aspekte mit Einfluss auf die modellierten Umweltauswirkungen, sind der Gebrauch von regionalspezifischen Arten-Abfluss Kurven sowie die Wahl der zur Bestimmung der Kurve verwendeten Regressionsfunktion. Obwohl diese Methode zur Wirkungsabschätzung ein hohes Potenzial für die Entwicklung von Charakterisierungsfaktoren für eine breite räumlich Abdeckung bietet, ist die Anwendung auf die Analyse von lokalen Wasserbewirtschaftungen nicht empfohlen (da möglicherweise eine höhere räumliche und zeitliche Auflösung zu einer adäquaten Abschätzung spezifischer Massnahmen zur Verringerung der Umweltauswirkungen nötig wäre). Hinzu kommt, dass diese Methode keinen kausalen Zusammenhang beweisen konnte. Die Abschätzung der Auswirkungen der Flusswasserentnahmen anhand eines deterministischen Wassertemperaturmodells erscheint im Gegensatz dazu besser auf die Entscheidungsprozesse in der Bewirtschaftung lokaler Wassereinzugsgebiete angepasst zu sein und baut zudem auf einem kausalen Zusammenhang auf. Allerdings limitieren die zur Modellierung benötigten detaillierten Daten den generellen Einsatz dieser Methode auf überregionaler Ebene (z.B. für die Ökobilanzierung) trotz des Versuchs der Vereinfachung der Berechnungen und der dazu benötigten Daten. Dass die Resultate dieses Modells sehr ortsspezifisch sind, zeigt auch das Ergebnis, dass im Fallstudiengebiet Änderungen der Flusstemperatur scheinbar weitaus geringere Auswirkungen zur Folge haben als eine Änderung der Abflussmenge.

Die Betrachtung der Umweltauswirkungen von Flusswasserentnahmen in der Ökobilanzierung landwirtschaftlicher Prozesse im Kontext des Klimawandels wird daher ausdrücklich empfohlen. Wichtig dabei ist es den Entnahmeort innerhalb eines Flusseinzugsgebiets und die Auswirkungen auf die aquatische Biodiversität zu berücksichtigen. Die Anpassung der Landwirtschaft an den Klimawandel sollte diverse Massnahmen kombiniert einsetzen, um der mehrfachen Zielsetzung gerecht zu werden (wie z.B. Ertragserhaltung, ökonomische Rentabilität und die Erhaltung einer intakten Umwelt). Die Konsequenzen regionaler Entscheidungen in einem breiteren nationalen und internationalen Kontext sollten, ebenso wie die sozialen Auswirkungen dieser Entscheidungen, untersucht werden um in der Zukunft ganzheitliche Nachhaltigkeitsanalysen für die Landwirtschaft im Kontext des Klimawandels zu ermöglichen.

RÉSUMÉ

Il est attendu que dans le siècle à venir, le changement climatique affectera l'agriculture à travers le monde, provoquant des baisses de productivité dans de nombreuses régions. En parallèle, une augmentation importante de la population mondiale est projetée. Ceci implique une pression accrue sur la production de nourriture, afin de garantir la sécurité alimentaire pour tous. En Suisse, des étés plus chauds et plus secs sont attendus d'ici 2050 dans les régions productives de plaine, ce qui pourrait amener à des baisses de rendement. Cependant, la Suisse a des objectifs d'autosuffisance partielle en alimentation, et a donc tout intérêt à maintenir ces rendements. Les agriculteurs souhaiteront également maintenir leur revenu à un niveau profitable. Ainsi l'agriculture Suisse doit s'adapter au changement climatique, à travers une modification de la gestion au niveau de la ferme, ainsi qu'un changement de la politique et des incitations au niveau régional et national. Cependant, de telles stratégies d'adaptation doivent également éviter l'augmentation des impacts environnementaux, si les objectifs environnementaux de la Suisse sont également à atteindre.

Cette thèse a ainsi évalué les impacts environnementaux de scénarios d'adaptation de l'agriculture au changement climatique pour deux cas d'étude en Suisse, au niveau de la ferme et de la région, à l'aide d'analyse de cycle de vie. Une option importante d'adaptation est l'utilisation d'eau de rivière pour l'irrigation, ce qui peut négativement affecter la biodiversité aquatique; cependant, aucune méthode satisfaisante n'était disponible pour évaluer de tels impacts. Une méthode adéquate d'évaluation des impacts, applicable à de larges échelles spatiales, a donc été développée dans cette thèse sur la base de la relation entre richesse en espèces et débit en rivière. Une méthode alternative d'évaluation des impacts a également été développée, qui modélise les changements de température de l'eau en rivière causés par les prélèvements d'eau, afin d'explorer un possible mécanisme d'impact spécifique. L'utilisation d'eau d'aquifère ainsi que l'augmentation de l'ombrage de la rivière ont de plus été évalués en tant qu'options alternatives de gestion des ressources en eau du bassin-versant.

Les résultats principaux montrent que si les agriculteurs s'adaptent d'un point de vue purement économique au changement climatique (comportement que l'on peut attendre en l'absence d'autres incitations), la productivité ainsi que l'efficacité environnementale diminuent à l'avenir. Ainsi il y a un besoin d'intervenir par des mesures politiques si l'on veut maintenir la productivité et l'efficacité environnementale dans une région. L'impact de l'utilisation d'eau de rivière sur la biodiversité aquatique constitue le principal conflit avec l'adaptation de l'agriculture Suisse au changement climatique; il est

ainsi recommandé que de tels impacts soient pris en compte lors de l'évaluation de la performance environnementale de l'agriculture sous un climat altéré. La perte potentielle d'espèces aquatiques dans l'un des deux cas d'étude atteint 8% des espèces du bassin-versant, en contradiction avec le bas indice de stress hydrologique observé dans la région. En général, d'autres impacts environnementaux (tels que le potentiel de réchauffement planétaire), rapportés à la quantité produite, diminuent à l'avenir tant que la productivité des systèmes agricoles est maximisée. La minimisation des impacts environnementaux absolus (non-rapportés à la productivité) est quant à elle liée à une baisse marquée de la productivité. Une telle baisse de productivité peut à son tour conduire à une augmentation des importations, et ainsi des impacts causés aux sites de production impliqués. Les résultats sont fortement influencés par de possibles choix politiques, autant que par le changement climatique en soi: ainsi, la politique a un grand potentiel pour influencer et atténuer les impacts du changement climatique sur la productivité agricole et l'environnement. Au niveau de la ferme, les impacts varient selon la région et le type de ferme, indiquant que les mesures politiques devront peut-être intervenir avec une résolution appropriée. Les mesures politiques ciblant spécifiquement la restriction de l'utilisation d'eau au niveau de la ferme sont certes efficaces pour protéger la biodiversité aquatique, mais provoquent une diminution de la productivité et ainsi une hausse des impacts relatifs à la productivité. Ainsi l'atténuation des impacts sur la biodiversité aquatique au niveau régional, tout en maintenant le niveau de productivité agricole, nécessitera des mesures de gestion complémentaires. Des mesures multiples seront sans doute nécessaires, puisqu'une seule mesure (telle que l'utilisation d'eau d'aquifère à la place d'eau de rivière, ou une ombrage accrue de la rivière) ne suffit pas à éviter la totalité des impacts attendus (d'autres options à considérer sont la restriction de l'utilisation d'eau, la limitation du besoin d'irrigation à travers le choix des cultures et la gestion de l'intensité, la renaturation du chenal d'écoulement, le développement d'abris pour la biodiversité, ainsi que l'utilisation d'eau du lac).

L'évaluation des impacts de l'utilisation d'eau de rivière sur la biodiversité aquatique, à l'aide de la relation entre richesse en espèces et débit en rivière, est influencée par la résolution spatiale utilisée: la considération de l'emplacement des extractions d'eau à l'intérieur d'un bassin versant (telle que proposée par cette thèse) affecte l'amplitude des impacts estimés de manière non-négligeable. Si la perte de biodiversité est comptabilisée de manière absolue (et non pas relative à la biodiversité totale présente), la considération de plus nombreux taxons est très pertinente. Cette thèse propose de mesurer la perte absolue de biodiversité en la pondérant par un indice de sa vulnérabilité: cette approche est recommandée afin de considérer la relation avec une référence absolue de perte de biodiversité, telle qu'une extinction globale d'espèces. D'autres éléments qui influencent l'amplitude des

impacts potentiels dans cette méthode sont l'utilisation de relations entre richesse en espèces-débit en rivière régionalisées, ainsi que le choix de la fonction de régression utilisée pour modéliser cette relation. Cette méthode d'évaluation des impacts a un grand potentiel pour le développement de facteurs de caractérisation pour de grandes étendues spatiales, cependant elle n'est pas recommandée pour des évaluations locales et la gestion locale de bassin-versant (puisque ceci peut nécessiter plus de détail spatial et temporel afin d'évaluer des mesures d'atténuation spécifiques) ; en outre, elle ne s'appuie pas sur une relation de causalité prouvée.

L'évaluation des impacts de l'utilisation d'eau de rivière sur la biodiversité aquatique, à l'aide d'un modèle mécanistique du changement de température de l'eau, est à l'inverse mieux adaptée au soutien de la prise de décision au niveau du bassin-versant local, et s'appuie sur un mécanisme de causalité. Le niveau de détail requis par la méthode n'est cependant pas considéré adapté pour couvrir une large étendue spatiale (telle que rencontrée en analyse de cycle de vie), malgré les efforts pour minimiser les besoins en données et en capacité informatique. Les résultats produits par cette méthode montrent que les impacts peuvent être très localisés. Les changements de température en rivière causés par l'utilisation d'eau pour l'irrigation ne provoquent que de moindres impacts sur la biodiversité aquatique, comparés aux impacts causés par les changements de débit en soi.

Ainsi, dans un contexte de changement climatique, la considération des impacts d'utilisation d'eau sur la biodiversité aquatique dans l'analyse de cycle de vie de l'agriculture est fortement recommandée, avec en particulier la considération de l'emplacement des prélèvements d'eau à l'intérieur d'un bassin versant. L'utilisation de méthodes plus génériques, disponibles à des résolutions spatiales plus grossières, peut être insuffisant pour refléter des impacts importants. L'adaptation agricole et la gestion de bassin-versant devraient prendre en considération de multiples critères environnementaux et être prêts à considérer des combinaisons de solutions complémentaires. A l'avenir enfin, les conséquences nationales et internationales de décisions à l'échelle régionale devraient également être évaluées, ainsi que les implications sociales des stratégies d'adaptation, afin de fournir une évaluation complète de durabilité.

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GLOSSARY

Absolute impacts: total impacts occurring in a region or over a given area (in opposition to relative impacts, which are related to a functional unit such as MJ digestible energy).

Adaptation: change in behavior or management in order to readjust and, ideally, optimize a system under new and different conditions.

Average characterization factor: in life cycle assessment, the average impact factor calculated over a wide span of initial conditions; typically used for non-linear flow-impact relationships and when the change is flow is not a marginal increment of the initial flow, and therefore the impacts cannot be assumed to be a linear extrapolation of an incremental change.

Characterization factor: in life cycle assessment, value expressing the relative impact due to an incremental change in a flow. When multiplied by a change in a flow, provides the estimate of resulting impacts.

Discharge: volumetric longitudinal flow of water in a river (typically in m^3/s).

Eco-efficiency: relationship of environmental impacts to economic benefit or production, for a product or process.

Eco-region: an area with relatively homogeneous ecological conditions, within which comparisons and assessments of biodiversity are meaningful.

Effect factor: provides the relationship between an incremental change in the environment and a change in impacts for a given environmental indicator (such as a change in pollutant concentration). Is used in combination with the fate factor to form the characterization factor.

Extent of occurrence: area within which a species has been observed to occur.

Fate factor: provides the relationship between an incremental change in a flow and a change in the environment (such as the fate of a pollutant in the environment and its final concentration). Is used in combination with the effect factor to form the characterization factor.

Flow: in life cycle assessment, a unit process, product, input or emission, that constitutes part of the inventory and causes environmental impacts.

Functional unit: in life cycle assessment, impacts are always related to a function of the product or process assessed (otherwise, minimizing impacts would lead to abandonment of the entire product or process). This function (e.g. production of apples) is measured by the functional unit (e.g. kg of apples); impacts are then expressed "per" functional unit.

Impact category: in life cycle assessment, a group of environmental impacts representing a particular area of concern, into which the impacts can be classified.

Inventory: list of all inputs (natural and man-made) required, and emissions caused, in the life cycle of a product or process. In combination with a life cycle impact assessment, forms a life cycle assessment.

Life cycle assessment: holistic environmental impact assessment of a product or process, listing inputs and emissions as well as potential environmental impacts throughout the entire life cycle of a process or product (i.e. from resource extraction through production and use, to waste disposal).

Life cycle impact assessment: part of a life cycle assessment that relates the flows identified in the inventory to potential impacts on the environment, usually through the use of characterization factors.

Longitudinal zonation: separation of a river network into distinct zones based on ecological and/or physical criteria, that form a longitudinal sequence along an upstream-downstream gradient of the river.

Marginal characterization factor: in life cycle assessment, the impact factor specific to an initial flow amount. Typically calculated as the derivative of the relationship between a flow and an impact, at a given initial flow amount. Used when the change in flow is a marginal increment of the initial flow amount, and thus the relationship between impact and flow can be locally approximated to linear.

Mitigation: avoidance or reduction of impacts caused by a change in a system through direct or complementary measures.

Optimization: mathematically, finding the best result of some objective function given a defined range of input variables. In an applied case, adjusting management and behavior in order to maximize an output and/or minimize an emission for a given set of inputs and management variables.

Principal component analysis: a statistical data exploration method, which identifies similarities in behavior between variables (e.g. impacts) for a set of observations (e.g. scenarios), enabling a reduction of the complexity and number of variables required to explain differences between observations.

Regression function: a mathematical expression used to represent the relationship between a variable and one or several predictors.

Relative impacts: impacts related to a functional unit such as MJ digestible energy (in opposition to absolute impacts, which are the total impacts occurring).

River temperature regime: description of the typical temperature time series of a river, including inter-annual, seasonal and daily temperature fluctuation cycles (can include quantitative terms such as minima, means and maxima for example).

Scenario: set of assumptions about a future situation.

Species richness: number (or count) of species in an ecosystem.

Species-discharge relationship: mathematical expression describing the observed behavior of aquatic species richness according to river discharge (typically established based on statistical regression of observations of species richness and discharge in rivers).

System boundary: limits between a system modeled in life cycle assessment, and the environment. This indicates which flows and processes are to be included in the life cycle assessment.

Taxon: group of species that form a recognized taxonomic unit (ideally, this represents a group with evolutionary or phylogenetic relationships such as a common ancestor, but is historically not always the case: for example fish, invertebrates, reptiles).

Threat status: evaluation of the extinction risk of a biological species.

Uncertainty: the state of having limited knowledge about a situation, often due to limitations in available information (such as data), simplifications in assumptions and models, and errors. The uncertainty of a result in life cycle assessment represents the range of plausible outcomes for that result around an expected value, based on uncertainty propagation of input values.

Water consumption: in general in life cycle assessment, indicates the evaporative use of water (use which effectively removes water from its relevant environment, through integration in a product or ultimate evaporation). In the case of river water abstraction, water consumption is the use of water abstracted from the river, which is not rapidly returned to practically the same location in the river.

Withdrawal (consumptive): abstraction of water from a water body (if consumptive, this abstraction is not rapidly returned to practically the same location in the river, thus effectively depriving a portion of the river of its water).

LIST OF ABBREVIATIONS

ABL:	Potential aquatic biodiversity loss
AIC:	Akaike information criterion
CF:	Characterization factor
CH:	Switzerland
CWF:	Cumulative Weibull Function
EF:	Effect factor
EPT:	Ephemera, plecoptera and trichoptera
EU:	European union
FAO:	Food and Agriculture Organization of the United Nations
FF:	Fate factor
FWE:	Freshwater eutrophication potential
GSE:	Global species extinction equivalents (weighted by vulnerability)
GWP:	Global warming potential
IUCN:	International Union for the Conservation of Nature
LCIA:	Life cycle impact assessment
NSC:	Nash-Sutcliffe coefficient
PDF:	Potentially disappeared fraction of species
RF:	Rarity factor
RMSE:	Root mean square error
SDR:	Species-discharge relationship
SR:	Species richness
TBR:	Terrestrial biodiversity reduction
TEP:	Terrestrial ecotoxicity potential
TS:	Threat status
WSI:	Water stress index

1 INTRODUCTION

1.1 Problem statement and literature review

Climate change is expected to affect Swiss agricultural practice in the next 50 years: indeed, increasing temperatures, decreased summer precipitation and intensified extreme weather events (including drought) may cause changes in yields and create water-related risks in Swiss agriculture [1]. It is expected that agriculture will be able to adapt at least partially to these new climatic conditions [2], with sufficient technology, capital and management modifications at the farm scale (decision unit of the farmer), at the regional scale (decision unit of local policy-makers) and at the national and international scale (decision scale of national policy-makers) [3]. However, from a sustainability perspective, it is important that these adaptation strategies not only ensure economic profitability and maintain productivity, but also avoid the deterioration of environmental conditions [4, 5]. Thus the trade-offs between the costs, benefits and impacts of adaptation should be assessed.

The impacts of climate change on agricultural productivity and profitability, and possible agricultural adaptation measures, have been studied for Switzerland in the context of the AGWAM project [6-9]. This project developed agricultural adaptation strategies for the climate in 2050 in Switzerland, at the farm and regional scale, which maximize profit and productivity while avoiding increases in environmental impacts. These adaptation strategies focus on farm and land use management options, involving spatially-explicit changes in land allocation to different crops, changes in fertilization, tillage and irrigation intensity, and changes in livestock numbers. The present thesis was embedded in the AGWAM project and addressed the extended environmental impacts of the adaptation strategies, which is necessary in order to identify trade-offs and support impact mitigation while adapting agriculture to climate change.

"Environmental impacts" is a broad term covering many different aspects, related to ecosystem quality, biodiversity, resource preservation, greenhouse gas emissions, to name but a few. It is essential to consider as many relevant indicators of environmental impacts as possible, in order to ensure that potential trade-offs between different aspects are captured and burden shifting is avoided [10]. Life cycle assessment (LCA) is a framework for assessment of the environmental impacts of a product, process or system, which considers the impacts of its entire "life cycle" (from resource extraction, through processing and consumption, to waste disposal). Multiple environmental indicators can be

addressed in order to reflect a multi-criteria view of the generic term "environmental impacts". LCA thus enables identification of burden shifting along the life cycle, and of trade-offs between environmental indicators. LCA has been found to be an adequate framework for assessing whole-farm environmental impacts [11]. The only existing operational methodology for farm LCA, including inventory and impact assessment tools as well as a database is SALCA [12], specifically developed for Switzerland. Further environmental impact assessment approaches developed for farms include REPRO [13] and RISE [14] and the approach of Eckert et al. (2000) [15], however these either do not consider the entire farm life cycle, or are less complete in terms of impact pathways considered [16]. The LCA of agricultural regions is an approach for which no conclusive methodology exists yet [17], with few and only very recent examples of application [18], again none for Switzerland. Many LCA studies have been conducted at the crop level for current conditions in Switzerland [19-21], and one important study exists at the farm level [22]. However, none have addressed the impacts of agriculture under future climatic conditions. In general, studies assessing the environmental impacts of agricultural scenarios under future climate, at the level of detail provided by LCA, do not seem to be available yet, although some reports provide general indications of the type of impacts that can be expected [23-25].

LCA of agricultural production systems typically use midpoint indicators, which can be modeled as direct impact pathways on a specific aspect of environmental impacts (such as global warming potential, eutrophication potential, toxicity etc.), in opposition to endpoint indicators, which involve further normalization and conversion of midpoint indicators to aggregated indicators representing areas of damage (ecosystem quality, resource depletion and human health). Indeed, midpoint indicators generally benefit from lower uncertainty and higher acceptance than endpoint indicators [26]. In the case of agriculture, they additionally have the benefit of representing impacts that stakeholders can easily understand and relate to. It is further recommended in the case of agricultural LCA to reduce the number of indicators used for communication of results, in order to simplify communication without losing information (on trade-offs for example) [22]. This has been done for current farm LCAs in Switzerland [22], but not for future scenarios nor for regional LCAs (in both cases, trade-offs and thus the selection of indicators may be different).

An important adaptation strategy for Swiss agriculture to climate change consists of an increased use of water for irrigation [27]. This may however cause competition with other water requirements, such as for human consumption (drinking water, cooling water, industrial processing, fisheries, and leisure) as well as for aquatic ecosystems. Competition with the latter use of water can result in potential impacts

on aquatic biodiversity in particular [28]. 71% of agricultural irrigation water is sourced from surface water [29], therefore the impacts of river water consumption in particular, including the impacts on river ecosystems themselves, should not be neglected in the LCA of agricultural adaptation strategies [30, 31].

In general, water use impact assessment is at an early development stage in LCA. Several methods have been proposed, and are reviewed by Kounina et al. (2012) [32]. Impact-oriented assessment methods for water consumption have been advocated in preference to inventory-related indicators, since they weight the consumption of water by a measure of exposure and response in a desired area of protection, and thus reflect the gravity of consuming a certain amount of water [33]. Life cycle impact assessment (LCIA) traditionally evaluates the impact of a process on an environmental indicator using characterization factors (CFs). These provide the unit change in the environmental indicator caused by a unit change in an impacting flow (such as water consumption). Characterization factors typically combine a fate factor with an effect factor. The first is defined here as providing the fate of a pollutant in the environment such as its final concentration, or the resulting change of a disturbance on the physical environment (e.g. change in amount of habitat available), whereas the second provides the effect of that change in concentration or in the physical environment on the environmental indicator itself [26, 34].

LCIA methods to assess damages to ecosystem quality due to water consumption include the water stress index approach by Pfister et al. (2009) [35], which uses the water-limited fraction of net primary productivity as an indicator of ecosystem damage and provides corresponding characterization factors at country and river basin resolution worldwide; the approach by Van Zelm et al. (2011) [36] which provides a characterization factor for the Netherlands, relating groundwater withdrawals to impacts on terrestrial vegetation through groundwater table lowering; and an index approach by Milà i Canals et al. (2009) [37] which estimates impacts on ecosystems by relating water use to water availability while accounting for ecosystems' water requirements. However none of these methods are entirely adequate in the context of water use for the case of Swiss agriculture under climate change scenarios, since they do not specifically reflect the most important impact pathway of river water consumption on aquatic biodiversity, are not available yet at the appropriate spatial resolution allowing comparison between regions in Switzerland (e.g. Pfister et al. (2009)), or are specific to another region (e.g. Van Zelm et al. (2011)). Characterization factors for the impact of river water consumption on freshwater biodiversity have been calculated by Hanafiah et al. (2011) [38]: the fate factor relates consumption (in m^3) to

reductions in river discharge as a one-to-one relationship (1 m^3/s withdrawn and consumed on average per year results in 1 m^3/s reduction in yearly average discharge at mouth). The effect factor is based on a relationship between the species richness of one taxa (fish) aggregated within river basins, to the average discharge at the mouth of the basins (the so-called species-discharge relationship or SDR, as developed for a dataset comprising a sample of basins distributed worldwide, by Xenopoulos et al. (2005) [39]). The final impact is measured as the potentially disappeared fraction of species (PDF) in the ecosystem, weighted by the volume of ecosystem affected for a certain time period, with units PDF*m^3*y. The method by Hanafiah et al. (2011) has several limitations for a direct application in Switzerland: the species-discharge relationship it uses was developed for latitudes below 42° and near-natural rivers, and hence its applicability is not verified for Switzerland and much of Europe (as well as Canada, much of Russia and Australia). The species-discharge relationship used was furthermore developed using basins across the world; higher precision in the species-discharge relationship might be achieved by developing species-discharge relationships specific to sub-regions. Approaches to model regionalized species-discharge relationships at a higher spatial resolution exist, including using sub-basins [40], river archetypes [41] and river reaches (e.g. for several regions in the USA [42]). However such regionalized species-discharge relationships are currently not available globally, nor for Switzerland and Europe. In addition, with a characterization factor calculated for entire river basins, local effects and influence of the location of withdrawal within the basin are not addressed: indeed, Hanafiah et al. (2011) estimate species loss in the entire watershed (using discharge at mouth), and weight by the river volume, regardless of location of the withdrawal. Finally, only the fish taxon is included. Although fish are commonly used and recommended as indicators of aquatic ecosystem health [43], using species richness of just one taxon remains a limitation [44]. Macro-invertebrates in particular could be more suitable indicators species for changes in hydrology in smaller streams [45] and generally exhibit lower mobility [46], and may thus be a more reliable indicator species of impacts due to lower migratory capability.

A further step beyond the assessment of environmental impacts related to water use is the identification of adequate mitigation options. Mitigation of environmental impacts may be best achieved through a holistic approach, making use of multiple management options [47].These include long-term strategic changes in agricultural practice such as those proposed in AGWAM (change in crop mix, land allocation, intensity of fertilization, tillage and irrigation, and livestock numbers), as well as technical options such as changes in crop varieties and irrigation technology. The latter may however be seen as incremental technological improvements, that may be insufficient to address long-term

changes of large magnitude. The mitigation of environmental impacts may also be enhanced by alternative management strategies of water resources. These should be developed at the relevant management scale, in this case the local watershed [47, 48]. Water use can be restricted using water pricing or quotas (as assessed in the AGWAM project [7]), or by allocating water withdrawals to another source: for example the alternative sourcing of river water withdrawals from groundwater [47] (surface and groundwater, as part of a whole system, should be jointly managed [49]). The impacts of groundwater use for agricultural irrigation, on terrestrial and aquatic ecosystems in Switzerland has not been addressed yet.

Additionally, the aggravating factors accompanying decreases in water availability due to water use can be addressed. For example, important changes in the river thermal regime can be mitigated by increased riparian shading [47]. Indeed, changes in the thermal regime of rivers due to climate change are also expected to cause significant impacts on aquatic ecosystems [50-52]. Changes in thermal regimes are significantly influenced not only directly by changes in air temperature, but also by changes in discharge [53, 54], in particular in situations where low flows coincide with high air temperatures [50]. Depending on the climate change scenario, Switzerland is expected to be exposed to a decrease in summer low flow and a simultaneous increase in water temperatures for the time horizon 2070-2100 [50]. Decreases in discharge can lead to a decrease in contaminant dilution and a decrease in habitat volume; the latter effect has been estimated to lead to decreases in local fish species richness of more than 10% for certain scenarios of climate change in the USA [39]. Increases in water temperature can additionally affect dissolved oxygen concentrations, modify chemical processes leading to an increased toxicity of pollutants, and exceed temperature tolerance ranges of certain aquatic species [50, 55, 56]. These effects can in turn affect the growth, composition and distribution of aquatic species [50, 57-60]. These issues have not been addressed in literature concerned with the adaptation of Swiss agriculture to climate change; estimates of the effects of climate change on aquatic ecosystems for Switzerland exist only for a single species of particular concern in Switzerland, the brown trout, for which the decrease in potential habitat due to climate change is assessed [61]. Estimates of potential aquatic biodiversity loss due to changes in river thermal regimes in Switzerland do not yet exist. Multiple models of river water temperature exist [62-69] but have yet to be operationalized for Switzerland; impact assessment methods relating changes in river water temperature to effects on aquatic biodiversity and ecosystems are on the other hand available for Switzerland [56, 70].

1.2 Objectives

This thesis aims to assess the relevant environmental implications of Swiss agricultural adaptation to climate change, explore potential impact mitigation strategies at the watershed scale, and generally improve methods to appropriately assess these impacts and strategies, particularly concerning aquatic biodiversity loss.

In particular, this thesis addresses the following issues:

- How can the impacts of river water consumption on aquatic biodiversity be assessed in LCA?
- What are the environmental impacts of agriculture in Switzerland under future climatic conditions?
- Which environmental indicators are most relevant in the context of agricultural adaptation to climate change, and which trade-offs emerge?
- Does adaptation of Swiss agriculture to climate change lead to reductions in its global warming potential, and therefore also contribute to climate change mitigation?
- To which extent can impacts of climate change and agricultural water use on aquatic biodiversity be mitigated at the watershed scale by water management strategies such as use of groundwater for irrigation, or riparian shading of the river system, and which are the most efficient strategies?

1.3 Approach in this thesis

After an introduction to the context of agricultural adaptation to climate change and the assessment of related environmental impacts (Chapter 1), this thesis presents the necessary methodological contributions developed for the LCA of agricultural adaptation strategies to climate change and the assessment of water management strategies (Chapters 2 to 3). It then applies these methods to two Swiss case studies by conducting an LCA of previously available agricultural adaptation scenarios (developed in the AGWAM project): the first addresses scenarios at the farm scale (relevant for farmers' decision-making) (Chapter 4). The second addresses scenarios at the watershed scale, and includes the evaluation of potential mitigation strategies for water-related impacts (Chapter 5).

In detail, Chapter 2 presents an adequate characterization factor model for the assessment of river water consumption impacts on aquatic biodiversity, applicable within an LCA framework. The issues mentioned in Chapter 1.1 are addressed in the following way: (1) provision of regionalized species-discharge relationships for Switzerland and Europe, for both fish and macro-invertebrates, and test of

several parameterizations and spatial resolutions (i.e. grouping watersheds per continent and per eco-region); (2) development of a refined LCIA method to calculate characterization factors, reflecting the location of consumption in the basin in order to verify its importance: this approach relies on longitudinal zonation of rivers within their watersheds, and uses global species extinction equivalents as a new indicator of biodiversity loss, enabling the weighting of potential species loss by a species rarity factor. This also provides a clear relation between local species loss and global species extinction; (3) sensitivity analysis of the results to the choice of model, parameterization and resolution. This approach is developed with the aim of allowing future extensions for worldwide application with a relatively moderate modeling effort, while maintaining as much detail and ecological relevance as possible.

However, it relies on a relationship based on correlation, not mechanistic causality. Therefore Chapter 3 develops an alternative approach to assess the potential impacts of river water consumption on aquatic biodiversity, explicitly modeling one causal impact pathway. This approach analyzes the impact pathway relating changes in river discharge to changes in aquatic biodiversity via changes in water temperature, using a deterministic model. This enables the verification of a mechanistic relationship, enhancing the "black box" approach followed in Chapter 2, and provides a more site-specific approach with an increased spatial resolution. It also provides the necessary fate factor modeling approach to assess the consequences of climate change, agricultural adaptation and riparian shading management on the river's thermal regime. Two existing models are then used to assess the effects of this change in water temperature on aquatic biota [56, 70]. The first provides an effect factor for the Rhine river (of which the case study region is a part) relating changes in ambient water temperature to the potentially disappeared fraction of aquatic species. The second uses a water temperature time series to score the ecological quality of the river, based on its deviation from the favourable and tolerable water temperature pattern for expected indicator fish species (identified based on the river type).

Chapter 4 presents an LCA of the farm-scale case study, including use of the method developed in Chapter 2. It evaluates the environmental impacts of farm adaptation to climate change from a purely economic perspective: this may be seen as the spontaneous adaptation farmers might implement if no other incentives are available. This chapter further analyzes the relevance of environmental indicators in the context of climate change, as well as the trade-offs that occur between environmental indicators, and with economic objectives. It includes an analysis of the sensitivity of outcomes to the choice of climate change scenario as well as future policy scenarios, and a discussion of the uncertainties related to assumptions for future scenarios (such as evolution of pesticide application, stakeholder acceptance,

and extrapolation to other regions). Based on this case study, the most relevant issues and case study region are identified for further detailed analysis at the watershed-scale in Chapter 5.

Chapter 5 presents the LCA of the watershed-scale case study for one region, with a focus on the mitigation of the cumulative regional impacts through regional measures. At this scale, the environmental impacts of potential policy strategies to address the shortcomings of the purely economic adaptation in Chapter 4 are addressed. The environmental impacts of the selected water management scenarios are also addressed in this chapter: riparian shading of the river (using the model developed in Chapter 3), and groundwater use for irrigation (a specifically developed groundwater model provides a fate factor, whereas the corresponding effect factor is drawn from literature [36]).

A general discussion is provided in Chapter 6, where the different approaches and spatial scales are compared, and their strengths and weaknesses discussed.

Chapter 7 concludes this thesis and proposes an outlook for further research and development.

2 LIFE CYCLE IMPACT ASSESSMENT METHOD FOR RIVER WATER CONSUMPTION

2.1 Introduction

Based on the literature overview concerning LCIA of water use and in particular river water consumption in Chapter 1.1, the research gaps identified include the lack of directly applicable methods for Switzerland, which relate river water consumption to an actual impact on aquatic biodiversity, and display a sufficient spatial resolution to reflect differences between watersheds in Switzerland and differences between locations within a watershed. There is also a lack of species-discharge relationships specifically developed for Europe and Switzerland and their eco-regions, as well as for taxa additional to fish. This chapter therefore addresses these gaps by developing region-specific species-discharge relationships at various scales (continent, country, and eco-region) for Europe and Switzerland, and including macro-invertebrate biodiversity. Furthermore, a new method assessing the impacts of river water consumption on aquatic biodiversity is provided, which considers the location of withdrawals within a river basin. The sensitivity of impact outcomes to the developments listed above is assessed using a Swiss case-study.

2.2 Method

2.2.1 Description

Figure 1 gives an overview of the assessment framework for the impacts of river water consumption on aquatic biodiversity. The new developments provided in this thesis are highlighted in red. The corresponding equations for each element are integrated in Figure 1, and the individual elements are explained in the following paragraphs.

$$ABL = \Delta_{W_j} \cdot CF_j \quad \textbf{(Eq. 1)}$$

*ABL : potential aquatic biodiversity loss, measured as vulnerability-weighted global species extinction equivalents [GSE*y]*
Δ_W : amount of water consumed [m³]
*CF : characterization factor [GSE*y/m³]*

Impacts are aggregated for all downstream zones i from withdrawal j to mouth of river.

$$CF_j = \sum_{i=j}^{mouth} (FF_i \cdot EF_i) \quad \textbf{(Eq. 2)}$$

FF_i : fate factor in zone i [m³/m³] (assumed to be 1 downstream of withdrawal, 0 upstream)
*EF_i : effect factor in zone i [GSE*y/m³]*

$$FF_i = \frac{d_{Q_i}}{d_{W_j}} \quad \textbf{(Eq. 3)}$$

d_Q : marginal change in discharge (due to change in water consumption rate) [m³/y]
d_W : marginal change in water consumption rate [m³/y]

$$EF_i = \frac{d_{SR_i}}{d_{Q_i}} \cdot \overline{RF}_i \cdot \overline{TF}_i \quad \textbf{(Eq. 4)}$$

d_{SR} : marginal change in species richness [species]
d_Q : marginal change in discharge [m³/y]
$\overline{RF}$: average species rarity weighting factor [m²/m²]
$\overline{TF}$: average species threat status weighting factor [-]

$$\overline{TF}_i = \frac{\sum_{s=1}^{SR_i} TS_s}{SR_{Q_{i,o}} \cdot TS_{max}} \quad \textbf{(Eq. 6)}$$

SR : species richness in zone i [species]
SR_{Q_o} : species richness at initial discharge of zone i [species]
TS_s : threat status of species s [-]
TS_{max} : maximum threat status (=5) as normalization factor [-]

$$\overline{RF}_i = \frac{\sum_{s=1}^{SR} \frac{A_{i,aff}}{A_{hab,s}}}{SR_{Q_{i,o}}} \quad \textbf{(Eq. 5)}$$

$A_{i,aff}$: area affected by withdrawal (= area of zone i) [m²]
$A_{hab,s}$: total habitat area of species s (≈ global extent of occurrence) [m²]

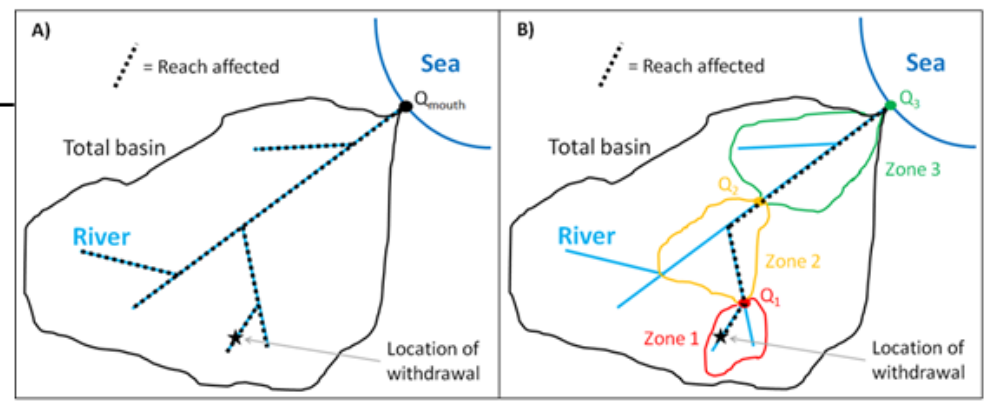
Species-discharge relationhsip (SDR):

- Taxa considered
  - Fish
  - Macro-invertebrates (EPT)
- Spatial scale
  - Continent (Europe)
  - Country (Switzerland)
  - Eco-regions
  - Longitudinal river zones: location of withdrawal in river basin can be reflected.
- Parameterization:

Power $$SR = aQ^b \quad \textbf{(Eq. 7)}$$

Cumulative Weibull $$SR = a\left(1 - e^{-bQ^c}\right) \quad \textbf{(Eq. 8)}$$

SR : species richness [species]
Q : discharge [m³/y]
a, b, c : regression parameters



A) Watershed-level SDR: species richness in total basin related to discharge at mouth Q_{mouth}.
B) Zone-level SDR: species richness in longitudinal river zones (e.g. Zone 1, zone 2 etc.) related to corresponding discharges (Q_1, Q_2 etc.). Withdrawal only affects downstream reaches.

Figure 1: *impact assessment framework and relevant equations. The novel components addressed in this thesis are highlighted in red.*

The calculation of potential impacts using a characterization factor is given by Eq. 1 (Figure 1). The characterization factor (composed of a fate and effect factor) is given in Eq. 2 (Figure 1). The fate factor (Figure 1, Eq. 3) is taken directly from Hanafiah et al. (2011) [38], and is assumed to be 1 for all parts of the river affected. The developments of this thesis principally address the effect factor (Figure 1, Eq. 4), where the potential change in species due to a change in discharge is calculated using the derivative of the species-discharge relationship, assuming marginal changes in discharge [71].

Hanafiah et al. (2011) used a species-discharge relationship that relates the discharge of a total river basin to species richness within the basin (count of unique species occurring within the basin) [39] . This approach is hereafter referred to as "watershed-level" (illustrated in Figure 1, Map A), and assumes that there is only one zone of river affected (= the entire watershed); thus i = 1 in Eq. 2 (Figure 1). The watershed-level approach was applied to Europe, and region-specific species-discharge relationships were developed for several eco-regions in Europe (according to the Water Framework Directive [72]: defined as an area with relatively homogeneous ecological conditions, within which comparisons and assessments of biodiversity are meaningful), as well as for Switzerland and the Swiss lowlands orographic region [73] (considered representative for the Swiss lowlands biogeographic eco-region [74]). These eco-regions are shown in Figure 2 and Figure 3; the corresponding species-discharge relationships are developed using only the watersheds they contain.

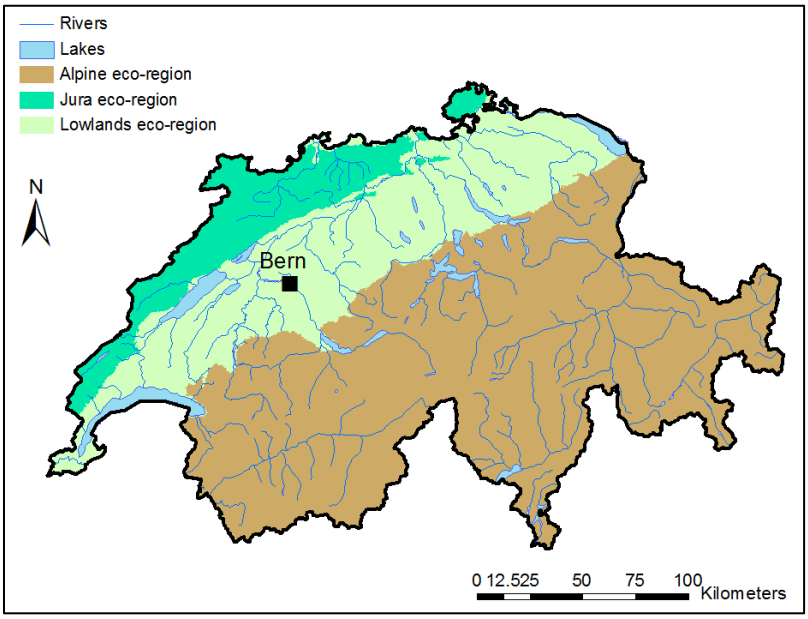


Figure 2: *eco-regions considered for Switzerland (adapted from Szerenczits et al. (2009) [73])*

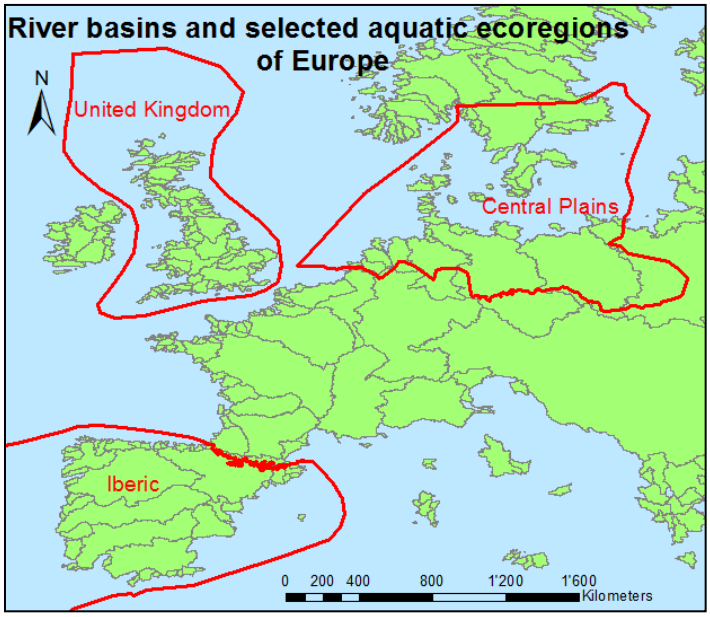


Figure 3: *the European river basins and eco-regions used (adapted from the Water Framework Directive (2000) [72] and CCM project (2007) [75]). Other eco-regions are not displayed here and were not used, since the number of data points available was too limited (using the species-discharge relationship developed for all of Europe for these regions is therefore recommended).*

Furthermore, watershed-level species-discharge relationships were developed for Switzerland for a subgroup of macro-invertebrates consisting of ephemera, plecoptera and trichoptera taxa (commonly referred to as EPT). EPT are generally regarded as sensitive to disturbances, and 62% of Swiss EPT species are considered threatened or near threatened according to the IUCN Red List criteria [76].

Species-area relationships and likewise species-discharge relationships are often assumed to follow a power regression function [77] (Figure 1, Eq. 7). This was used here as default regression function. However applying this function to latitudes above 42° or non-natural rivers may overestimate species richness, due to an asymptotical behavior in the species-discharge relationship at these latitudes [38] or within disturbed rivers (the species-discharge relationship curve flattens out at large discharges, suggesting a maximum limit in species richness). This is relevant for Switzerland and Europe, since actual maximum species richness is lower than predicted by SDRs developed for basins distributed worldwide (this could be due to recent glaciations [78] or high human disturbance levels). Therefore other regression functions for species-area relationships suggested in literature [79] were tested for the example of Switzerland. In particular, the cumulative Weibull function (Figure 1, Eq. 8) was retained as a possible alternative and applied for the Swiss species-discharge relationships, since it can simulate an asymptote.

In order to increase spatial detail and account for the location of water consumption in the basin, another approach for developing species-discharge relationships was applied (illustrated in Figure 1, Map B): species richness is counted in distinct longitudinal zones (subdivisions of the whole basin), and is related to the discharge of each zone, providing what is hereafter referred to as a "zone-level" species-discharge relationship [42, 54, 80]. Longitudinal zones are in essence defined to distinguish different species assemblages, with the assumption that each zone contains mainly different species (discussed in Chapter 2.3.4). Zonation may vary according to region and taxon [81]. This approach allows aggregation of the downstream effects from the point of withdrawal to the river mouth: the loss of species in each zone affected by the withdrawal can be summed, without double-counting of species (assumed to be distinct in each zone). A zone-level species-discharge relationship was developed for fish in Switzerland, using a longitudinal zonation proposed for Europe [81, 82] which defines four fish zones (trout, grayling, barbel and bream) based on the slope and width of the river. All regression modeling and statistical tests were performed using the statistical software package "R" [83]. Regression fitting objective was maximizing Pearson's R^2.

The species loss estimated according to the zone-level approach is assumed to affect the river system downstream of the withdrawal; however, this "local" loss does not inform us on the gravity of the impact from a global perspective. Therefore the local species loss in a zone is weighted by the ratio of the area affected (= area of zone) to the total global habitat area for each species occurring in the zone (in a similar way to previous work) [84]. The total habitat area is estimated from global occurrence points (see Table 1 for data sources) in a very simple way using GIS (illustrated in Figure 4): cells of a raster grid are attributed a non-null value if they contain at least one occurrence point (the resolution used here is 1 arc-degree; this is a coarse resolution but can be chosen differently for subsequent applications). Thus the probable extent of habitat area per species is the summed area of all non-null cells (regardless of the density of points occurring in each cell). Consistently, the same approach is applied to the river segments which constitute a zone, by summing the area of cells (with identical resolution, in this case 1 arc-degree) which the segments traverse.

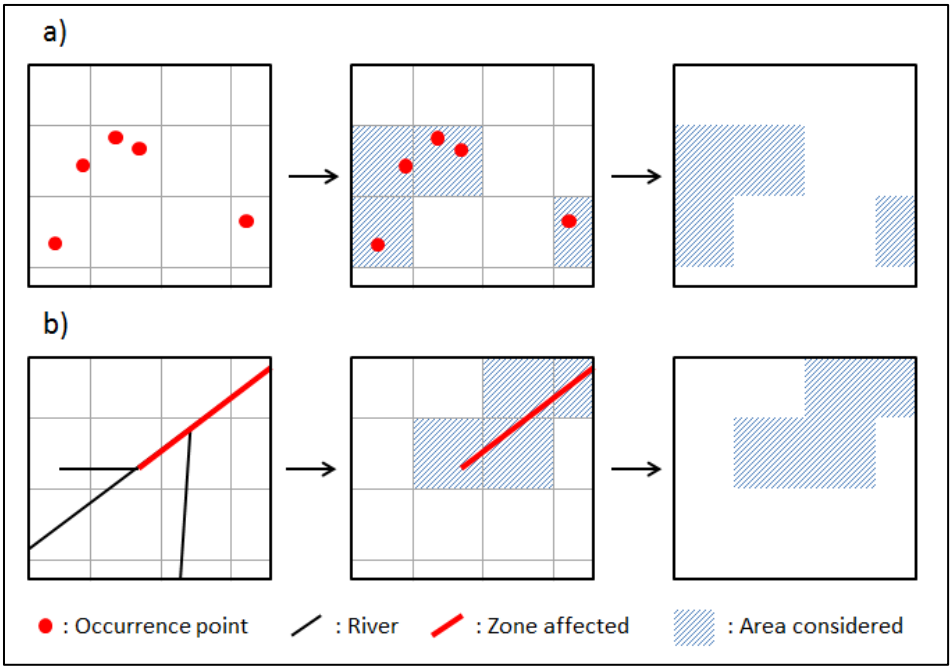


Figure 4: *a) schematic representation of the estimation of habitat area from occurrence point data; b) schematic representation of the estimation of affected area from river segment data.*

This converts local species loss into global species extinction equivalents, making local losses comparable to each other. Since the approach here does not define precisely which species are lost, the average rarity of all the species present in a zone is used (Figure 1, Eq. 5).

In addition to weighting the species loss by species rarity, it is further weighted by the normalized extinction threat status of the species in that zone [84] (based on the IUCN Red List [85]), also averaged for all species within a zone (Figure 1, Eq. 6). The IUCN threat status of species occurring in the wild reflects further vulnerability of species by considering multiple criteria such as abundance, turnover rate, fragmentation, dynamics in geographic extent etc. [86]. The IUCN threat status is qualitative, and was converted to a quantitative value, assuming a linear increment in vulnerability (as is often used per default in ecology [87]). The threat status is thus interpreted here as a scale of one to five, one being the category "least concern" and five being the category "critically endangered" (conversion in Table A1.1). Thus the loss of a critically endangered species is weighted five times higher than the loss of a species of least concern (note that this scale may be adapted if justified in future). The maximum threat status (= 5) is used as a normalization factor. Together, the rarity factor and the threat status factor provide a weighting of species loss by an indicator of their vulnerability. The reference state is the current extent of occurrence and threat status of the species concerned (rather than a historical natural state, since

many rivers are by far no longer natural, and some species are already extinct); this reference should be updated regularly, when the extents of occurrence and threat status data are updated.

Finally, using the zone-level approach, the marginal characterization factor providing the impact on aquatic biodiversity for a consumptive withdrawal of river water in a river zone j is given by Eq. 2 (Figure 1), and includes impacts in all the subsequent downstream zones i from the withdrawal to the river mouth. The fate factor dQ/dW, showing change in discharge due to a change in consumption rate dW, is assumed to be 1 in the river zone of withdrawal as well as in all zones downstream (and 0 in all upstream zones).

Note that this approach differs from the previously used PDF*m^3*y: vulnerability-weighted equivalents of global extinction [GSE*y] are taken here as an indicator of potential impacts on biodiversity, in the form of biodiversity loss. This uses the absolute number of species potentially lost (weighted by the fraction of their habitat affected), which implies that an ecosystem for which a higher amount of taxa has been considered will in consequence probably show a higher impact. A normalization of different taxa, for example by the characteristic species richness of the taxa, may be a way of addressing this issue (Eq. 9):

$$\Delta SR_{weighted} = \frac{\sum_{i=taxa\,1}^{NT}\left(\frac{\Delta SR_i}{SR_i}\right)}{NT} \qquad (9)$$

Where $\Delta SR_{weighted}$ is the weighted total potential species loss, NT is the number of taxa considered, ΔSR_i is the potential species loss in taxa i, and SR_i is the total species richness of taxa i (known or estimated). This provides the average fraction of taxa species lost (different from PDF, which provides the fraction of species lost in a local ecosystem).

If assuming non-marginal changes, an average characterization factor can be used (indeed, the species-discharge relationships used are not linear over large ranges of discharge): the species loss per unit of discharge reduction is estimated as the slope between the original species richness and zero (rather than the slope of the species-discharge relationship derivative at the original discharge, as for the marginal characterization factor), and is therefore the average species loss over the entire discharge available (Eq. 10).

$$CF_{nm,j} = \sum_{i=j}^{mouth}\left(\frac{SR_{Q_{0,i}}}{Q_{0,i}} RF_i \cdot TF_i\right) \qquad (10)$$

Where $CF_{nm,j}$ is the average characterization factor for a non-marginal withdrawal in zone j, aggregating impacts on all subsequent downstream zones which are also non-marginally affected (in GSE * y/m^3), $SR_{Qo,i}$ is the original species richness in zone i predicted using the species-discharge relationship with original discharge $Q_{o,i}$ in zone i, and RF_i and TF_i are the zone rarity and threat factors respectively for zone i. This characterization factor always gives a more extreme estimate than the marginal characterization factor.

2.2.2 Data sources

The sources of all data used are listed in Table 1.

Table 1: *data sources*

Region	Switzerland	Europe	Global
Parameter			
Species occurrence (for habitat area)	Swiss Center for Biological Records [88]		Global Biodiversity Information Facility [89]
Discharge	Swiss Federal Office for the Environment [90]	Global Runoff Database [91], European Water Archive [92]	
Catchment delimitation	Swiss Federal Office for the Environment [93]	Joint Research Center [94]	
Eco-regions	Agroscope [73]	Water Framework Directive [72]	
River width	Swiss Federal Office for the Environment [95]		
Slope	Swiss Federal Office of Topography [96]	European Environmental Agency [97]	

2.2.3 Sensitivity analysis

In order to assess the sensitivity of results to these developments, a case study example was conducted. This consists of the regional agricultural adaptation scenario "Ext2050, productivity" (see Chapter 5.1.3 for details). This scenario maximizes yields in the region for the climate in 2050, and implies a drastic increase in irrigation water demand, from 1.13 mio m^3/y currently to 46.24 mio m^3/y in 2050. Assuming an efficiency of 70%, the resulting river water consumption is 32.23 mio m^3/y (equivalent to a decrease in average discharge of 1.02 m^3/s). This is roughly 9% of the Broye river discharge (initial yearly average discharge 11.73 m^3/s) and 0.05% of the total Rhine river discharge (initial yearly average discharge

2254.06 m^3/s). The characterization factor method and zone-level species-discharge relationship for fish developed here were used to calculate the potential impacts on aquatic biodiversity.

The sensitivity of the results to the following choices was assessed: (1) location of the withdrawal: using the method proposed here and the zone-level SDR developed for Switzerland, impacts were calculated in GSE*y according to two hypothetical locations of withdrawal (shown in Figure 5); (2) use of region-specific species-discharge relationships for fish at different geographical scales (non-regionalized, Europe, Switzerland, Swiss Lowlands): impacts were calculated both according to the method proposed here, and the method by Hanafiah et al. (2011), assuming the withdrawal occurs in the Broye catchment (see Figure 5). Species vulnerability weighting when using the method developed here is the same no matter which SDR is used. This allows an estimation of the uncertainty in the result if a sub-optimal SDR is used; (3) inclusion of an additional taxon (EPT), providing SDRs for both EPT alone, and for total fish and EPT species richness combined. Only vulnerability weighting as developed for fish could however be applied for impact calculation as GSE*y (this was not available for EPT); (4) use of the cumulative Weibull function SDR (weighting otherwise the same as above).

The location of the Broye within the Rhine basin, the assumed locations of the consumptive withdrawals, and the delimitation of the longitudinal zones used in the case study are shown in Figure 5 (other tributaries of the Rhine would likewise be attributed with a similar zonation; here, only the zonation strictly downstream of the assumed withdrawal locations is represented along with the upstream trout zone).

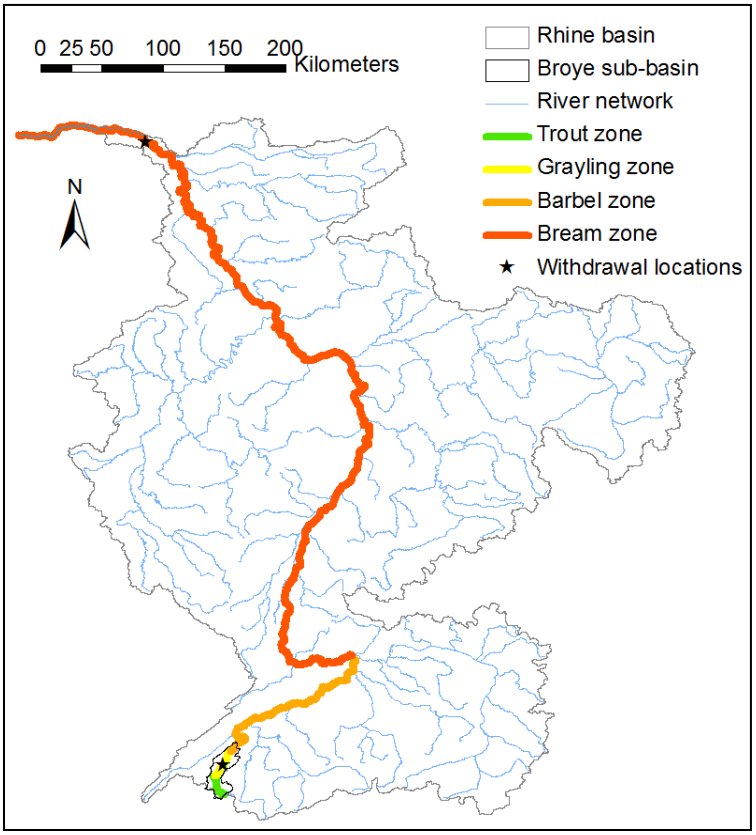


Figure 5: *map of Rhine basin with Broye sub-catchment, river network, assumed locations of consumptive withdrawals, and longitudinal fish assemblage zones for case study (based on Huet et al.(1949) [82]): trout, grayling, barbel and bream.*

2.3 Results and discussion

2.3.1 Species-discharge relationships

Table 2 provides the equations of all the watershed-level species-discharge relationships developed along with R^2 as a measure of goodness of fit, and sample size n (graphical representations of the principal species-discharge relationships are available in Figure 8, with detailed representations of the Swiss SDRs in Figure 6 and of the European SDRS in Figure 7).

Table 2: *equations of the watershed-level species-discharge relationships developed (y = species richness, x = average discharge in m^3/s), with R^2 as a measure of goodness of fit, and sample size n.*

SDR (region, taxa, regression function)	Equation	R^2	n	Figure
Europe, fish, power	$y = 7.82x^{0.30}$	0.35	271	7a, 8a
Central Plains, fish, power	$y = 18.75x^{0.15}$	0.39	29	7b

Iberic, fish, power	$y = 6.92x^{0.28}$	0.59	25	7c
United Kingdom, fish, power	$y = 15.93x^{0.06}$	0.04	104	7d
Europe, fish, CWF	$y = 6211.01(1 - e^{-0.001x^{0.30}})$	0.35	271	-
Switzerland, fish, power	$y = 2.84x^{0.45}$	0.42	662	8a
Swiss Lowlands, fish, power	$y = 9.31x^{0.28}$	0.70	145	-
Swiss Lowlands, fish, CWF	$y = 47.13(1 - e^{-0.20x^{0.47}})$	0.73	145	6c, 8a
Switzerland, fish & EPT, power	$y = 22.54x^{0.43}$	0.69	582	6a
Swiss Lowlands, fish & EPT, power	$y = 45.07x^{0.34}$	0.90	121	6a, 6b
Swiss Lowlands, fish & EPT, CWF	$y = 326.7(1 - e^{-0.12x^{0.56}})$	0.93	121	6b
Swiss Lowlands, EPT, power	$y = 35.98x^{0.35}$	0.89	121	-
Swiss Lowlands, EPT, CWF	$y = 280.9(1 - e^{-0.11x^{0.57}})$	0.92	121	6c
Zone-level Switzerland – all zones, fish, power	$y = 9.32x^{0.21}$	0.53	53	8a, 8c
Zone-level Switzerland – trout zones, fish, power	$y = 7.75x^{0.20}$	0.21	35	8c
Zone-level Switzerland – grayling zones, fish, power	$y = 10.88x^{0.19}$	0.79	11	8c
Zone-level Switzerland – barbel zones, fish, power	$y = 19.26x^{0.08}$	0.46	5	8c
Zone-level Switzerland – bream zones, fish, power	$y = 10.63x^{0.20}$	0.95	3	8c

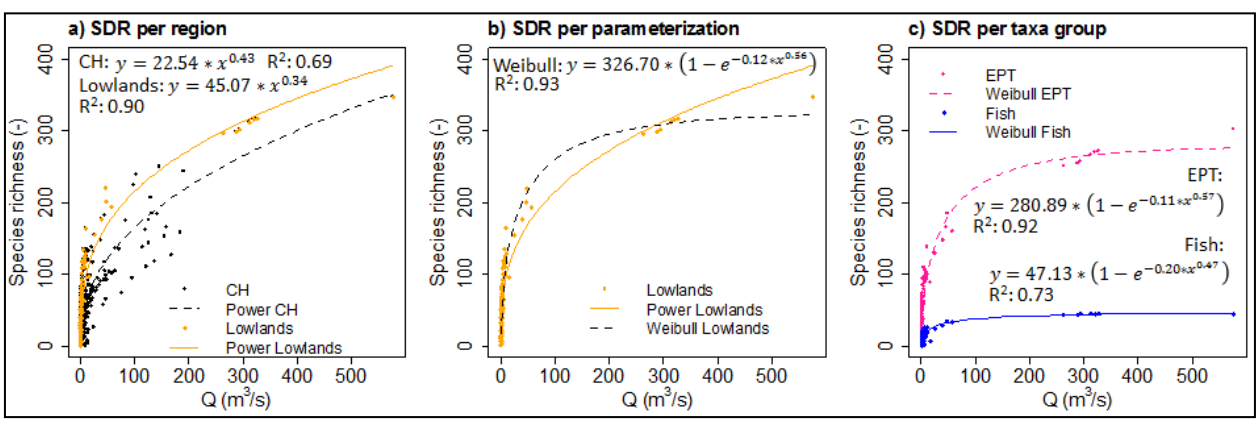


Figure 6: *watershed-level species-discharge relationships for Switzerland: a) Total species richness (fish and EPT) vs. annual average discharge, with distinction of all watersheds (black) and lowland eco-region*

watersheds (orange), power regression; b) Total species richness (fish and EPT added) vs. annual average discharge, lowland watersheds, with distinction of power (orange) and Weibull (black) regressions; c) Distinction of taxon group: fish (blue) and EPT (pink), for lowland watersheds and Weibull regression.

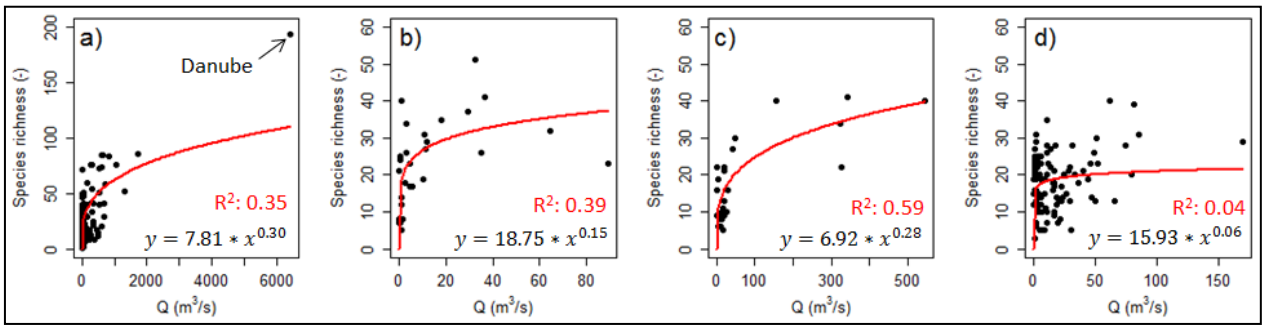


Figure 7: *species-discharge relationships for Europe and a selection of eco-regions: a) whole of Europe, b) Central Plains, c) Iberic, d) United Kingdom.*

All the species-discharge relationships show positive exponents as expected [80], suggesting that fish and EPT species richness in these regions would also be negatively affected by reductions in discharge. These results generally compare well with reference values in literature: the exponents of the power functions in Table 2 range from 0.06 to 0.45; literature provides an average exponent of 0.24 for a worldwide range of comparable aquatic and terrestrial species-area relationships [98], and the non-regionalized fish species-discharge relationship has an exponent of 0.4 [39]. The range of exponents found shows a high variability according to the region: similar variability has also been found for the USA, with exponents ranging from 0.02 to 0.196 [40]: this suggests that region-specific species-discharge relationships are preferable to average species-discharge relationships. The exponent of 0.30 found here for Europe (95% confidence interval = [0.25; 0.35]) is higher than a previous value of 0.23 for western Europe [99], which may be due to different data samples, or to changes in conditions. Higher exponents can indicate a stronger association between species richness and discharge, which in turn can indicate a higher vulnerability to changes in discharge [40, 100] and potentially more rapid extinctions [42]. The exponents for the Swiss species-discharge relationships are generally higher than for the European species-discharge relationships; the exponents for the Swiss Lowland species-discharge relationships are generally lower than for the whole of Switzerland; and the exponents for the EPT species-discharge relationships are higher than for the fish species-discharge relationships (suggesting that they may be more vulnerable to changes in discharge than fish, as has also been suggested for mussels) [40]. A notable outlier in the species-discharge relationship exponents is the case of the United Kingdom: this region shows a weak species-discharge relationship with the lowest exponent of

0.06 (similar to that found in certain regions of the USA) [40]. This may be due to the perturbing effect of other drivers of aquatic biodiversity, historical biogeographic constraints [40], or an unfavourable sampling (with very few data points for large discharges).

Where tested, the cumulative Weibull function provides a better fit than the power function, as measured by Pearson's R^2 (Table 2) and confirmed using the AIC (Swiss Lowlands, fish: AIC = 930 for power, 918 for CWF; Swiss Lowlands, fish and EPT: AIC = 1119 for power, 1075 for CWF; Swiss Lowlands, EPT: AIC = 1097 for power, 1061 for CWF). This suggests that the CWF may indeed be preferable for modeling species-discharge relationships in latitudes above 42° and other cases where the species-discharge relationship tends towards a limited maximum number of species.

Figure 8 compares a selection of the species-discharge relationships developed here with the species-discharge relationship used by Hanafiah et al. (2011) and shows details of the zone-level species-discharge relationship.

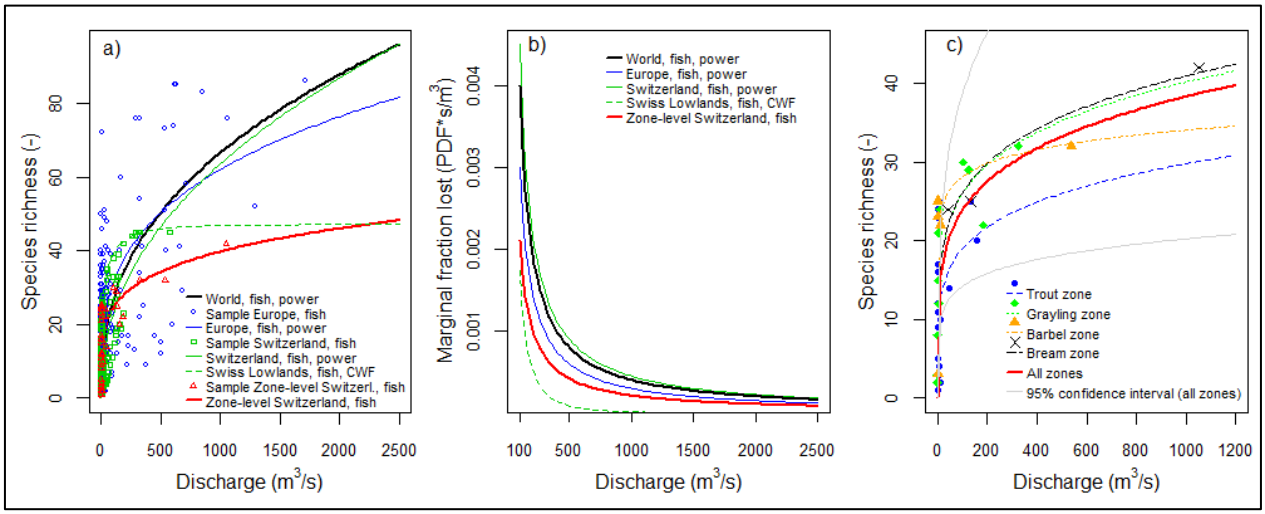


Figure 8: *a) comparison of the main SDRs developed here and the non-regionalized SDR [39], and overview of data samples used for each SDR; b) comparison of marginal species loss (PDF*s) per unit withdrawal (m^3) for these different SDRs, according to initial discharge (m^3/s); c) zone-level SDRs for fish in Switzerland, for all longitudinal fish zones combined ("all zones"), and with distinction of the four zones used ("trout", "grayling", "barbel", "bream").*

According to Figure 8a, the regionalization of the species-discharge relationship (non-regionalized – Europe – Switzerland) causes smaller differences in species richness prediction than changing the function used (e.g. CWF rather than power function). The asymptotical behavior of the species-

discharge relationship in the Swiss Lowlands can be seen through the shape of the species-discharge relationship using the CWF. The consequence of this behavior is that marginal changes in discharge no longer affect the predicted species richness above a threshold discharge. As previously mentioned, recent glaciations, or degraded quality of the rivers could explain this behavior. The species-discharge relationship at the zone-level shows a similar shape to the species-discharge relationships at the watershed-level, although predicted species richness is larger at the watershed-level (as can be expected, since species are aggregated over all zones within the whole watershed). Figure 8b shows that differences in predictions of marginal fraction of species lost (PDF*s/m^3) are maximal for rivers with lower discharges (e.g. 2.76E-3 PDF*s/m^3 at 100 m^3/s), however this rapidly decreases with increasing discharge, and the difference is much smaller for large rivers (e.g. 8.13E-4 PDF*s/m^3 at 500 m^3/s and 1.79E-4 PDF*s/m^3 at 2500 m^3/s).

The zone-level species-discharge relationship of fish for Switzerland was developed both per zone, and for all longitudinal fish zones combined (i.e. all data points together with no distinction of zones) (Figure 8c, equations in Table 2). Although it may be expected that the species-discharge relationships per zone are more precise (according to the hypothesis that species-discharge relationships for more homogeneous regions should show a better fit), the confidence intervals of the species-discharge relationships per zones were very large due to smaller data samples; the species-discharge relationship for all zones combined showed the smallest interval (and encompassed all the species-discharge relationships per zone at the 95% confidence level, with regression parameters a = [7.07; 11.63] and b = [0.15; 0.26]). Therefore the species-discharge relationship for all zones combined was used ($y = 9.32x^{0.21}$; R^2 = 0.53), which represents the general relationship between zone discharge and species richness, without distinction between zone type. Zone-level species-discharge relationships in various regions of the USA show exponents ranging from 0.13 to 0.45 [42, 80, 100], and are generally higher than watershed-level species-discharge relationships for the same regions. The zone-level species-discharge relationship exponent found here (= 0.21) fits within this range, although it's lower than that of the corresponding watershed-level species-discharge relationship (i.e. 0.45). The difference in species richness prediction at discharges of 1200 m^3/s reaches 12 species (between the trout and bream zone species-discharge relationships). It was verified that the area of each zone observed (each point in Figure 8c) is unrelated to its species richness (R^2 = 0.005, n=53; data available in Table A1.2). This supports the hypothesis that species richness is related to discharge rather than observation area in Switzerland.

2.3.2 Case study results and characterization factor

The rarity and threat status factors of the species concerned in each zone of the case study are given in Table A1.3. The potential impacts of the total river water consumption modeled for the case study amount to 1.80E-5 GSE*y. The relative contribution of each zone affected is provided in Table 3.

Table 3: *impacts and weighting factors for each zone affected in the case study, contributing to the total impact.*

Zone	Impact in zone [GSE*y]	Average threat factor	Average rarity factor	Species loss in zone [-]
Grayling	9.12E-06	0.22	1.12E-04	3.95E-1
Barbel	1.71E-06	0.20	6.82E-04	1.25E-2
Bream	7.18E-06	0.25	6.72E-03	4.22E-3

The contribution of each zone is not proportional to its size. Indeed, the proposed method captures an important feature of the species-discharge relationship by using several zones with their respective discharges, rather than using only the entire basin with the discharge at the mouth: the species-discharge relationship curve is steeper at smaller discharges, indicating that smaller sub-catchments are affected more severely than large sub-catchments by an identical withdrawal: indeed, species loss in the smaller zones (Grayling, Barbel) is higher than in the larger downstream zone (Bream). Other elements which influence the difference in impacts between the zones are the threat status of the species occurring there (in this case study, there is no large difference between the zones), and the habitat rarity factor. The rarity factor will tend to be higher for larger zones as shown in this case study, but also depends on the extent of occurrence of the species in each zone. Thus there is a certain compensation between species loss and rarity factor observed in Table 3.

The contribution of the threat status weighting in this case study is small, since all zones show a similar and low threat status. Removing the threat status factor in the case study (but keeping the normalization factor of 1/5 in order to preserve comparability) leads to a total potential impact of 1.56E-5, which is only a factor 0.8 lower than when the threat status factor is included. However, this factor is expected to show major differences for hotspots of biodiversity where many threatened species are located.

If the same withdrawal were to occur further downstream (for example in the agricultural plains of the Netherlands, see Figure 5), this would result in a potential loss of 7.18E-6 GSE*y: a factor 0.4 of the impact if withdrawn upstream. This difference is explained by two elements captured by the method proposed here: (1) in the first case (i.e. withdrawal in the upstream sub-catchment Broye), the potential impacts of the withdrawal in the first zones affected are high (the species-discharge relationship is steeper at lower discharges: rivers with smaller discharge have a higher loss of species per unit withdrawal) (see Table 3). In the second case (i.e. withdrawal in the agricultural plains of the Netherlands), the species-discharge relationship is flatter for the larger initial discharge. (2) The number of zones affected in the first case (withdrawal upstream) is higher than in the second case (withdrawal downstream), for which only the bream zone is considered affected. This difference would not be captured if using a method at the watershed-level, which would apply a single characterization factor for the whole of the Rhine basin. The sensitivity of dSR/dQ amongst basins worldwide using the dataset and non-regionalized species-discharge relationship provided by Hanafiah et al. (2011) was compared to the variability within the case study basin using the zone-level species-discharge relationship: for basins worldwide, dSR/dQ spans 3 orders of magnitude (1.1E-3 to 1.45 species*y/m^3) whereas within the case study basin, dSR/dQ spans 2 orders of magnitude (4.1E-3 to 5.8E-1 species*y/m^3). Thus the case study indicates that it may also important to consider intra-basin location, although this should be verified for other cases. The uncertainties assessed for the method developed in this thesis are smaller than this variability: using the 0.95 confidence interval of the zone-level species-discharge relationship, the calculated impact is a factor 2.2 lower or higher; as discussed in the next paragraph, the error if using an inadequate species-discharge relationship reaches a factor 2.6.

Note that relating potential local species loss to an equivalent of global extinction generally results in very small values for a marginal withdrawal. This does not mean that the impact is insignificant, since the potential species loss within the local basin or zone may still be high; additionally, large withdrawals may nevertheless result in large impact values.

2.3.3 Sensitivity analysis

The potential impacts calculated for the case study, using different species-discharge relationships and different characterization factor methods (as proposed here [GSE*y]; and according to Hanafiah et al. (2011) [PDF*m^3*y]) are shown in Table 4. Impacts in GSE*y are all calculated using the same vulnerability weighting; only the SDR used varies, showing the expected error if an inadequate SDR is

used. Impacts in PDF*m^3*y have no weighting (according to Hanafiah et al. (2011)), and are calculated using different SDRs.

Table 4: *impacts estimated for the case study, using different species-discharge relationships and characterization factor methods. Impacts in GSE*y are calculated according to the method presented in Figure 1, use the same zonation and vulnerability weighting in all cases and always assume the same location of withdrawal in the Broye region. Impacts in PDF*m^3*y are calculated for the case study, according to the method by Hanafiah et al. (2011) [38] but using various SDRs developed here in addition to the available non-regionalized SDR [78]. Equations used are available in Table 2.*

Row	*SDR*	*GSE*y*	*PDF*m^3*y*
1	Zone-level Switzerland, fish, power	1.80E-05	-
2	Non-regionalized, fish, power	4.57E-05	4.84E+04
3	Europe, fish, power	3.58E-05	3.63E+04
4	Switzerland, fish, power	4.75E-05	5.45E+04
5	Swiss Lowlands, fish, power	3.58E-05	3.39E+04
6	Switzerland, fish & EPT, power	-	5.21E+04
7	Swiss Lowlands, fish, CWF	2.16E-05	2.25E+02

The comparison of values in GSE*y and values in PDF*m^3*y is not directly possible. An interpretation of conversion between the two units is provided in Annex A1.5.

The characterization factor proposed here (in GSE*y/m^3) can be calculated using watershed-level species-discharge relationships such as the non-regionalized species-discharge relationship [39], rather than a specifically developed zone-level species-discharge relationship (e.g. if the latter is unavailable). However this overestimates the potential impacts as GSE*y by a factor ~2.6 (Table 4, rows 1 and 4), reflecting the double-counting of species loss mentioned in the methods section. In general, all impacts in GSE*y using a watershed-level SDR rather than a zone-level SDR results in some overestimation of impacts due to partial double-counting (impacts are aggregated in all zones downstream, although the SDR used to estimate the loss in each zone is developed for the watershed-scale with no distinction of species in zones).

The choice of region-specific species-discharge relationship causes differences in results of a factor 1.3 (as GSE*y), respectively a factor 1.6 (as PDF*m^3*y) (Table 4, rows 2, 3, 4, and 5). Using the non-regionalized species-discharge relationship (e.g. if a region-specific species-discharge relationship is not available) could therefore over- or underestimate the impact, depending on the region.

The inclusion of a further taxon (EPT) could not be estimated as GSE*y because the global extent of occurrence of these species was not available. Nevertheless, it increases the absolute species loss by a factor 7.2 (from 0.47 to 3.37 species). Indeed using a species-discharge relationship for both fish and EPT species results in a higher total species loss, since it sums the loss of EPT species as well as fish species. This confirms that comparison of biodiversity loss estimates between different ecosystems (e.g. terrestrial and aquatic) would require a normalization of the different taxa considered. The weighting suggested in Eq. 9 could be used for this purpose, but is optional: impacts may be kept in the original GSE*y; it is also possible to keep the impacts on different taxa separate rather than aggregating them. The ISO norm 14044 for life cycle assessment requests that in case of weighting, original values prior to weighting should be provided alongside weighted values. If using relative species loss (as PDF*m^3*y), the difference in potential impact when including EPT is less relevant (a factor 1.5 in this case): thus, when assessing relative species loss, fish alone may be supposed to be a plausible indicator also for EPT.

Potential impacts using the cumulative Weibull function were lower in the case study than if using the power function (Table 4, rows 5 and 7), due to the asymptotical behavior of the CWF (in particular if considering relative species loss as PDF). However due to the restricted number of observations for large discharges, the assumption that the species-discharge relationship in Switzerland truly follows an asymptotical behavior remains uncertain.

In the proposed method, the rarity factor and threat status are assumed to reflect different elements. The IUCN threat status reflects change of habitat extent, but not the actual extent itself. The two weighting factors were averaged individually for each zone in this thesis and then multiplied; they could also be multiplied for each species individually, and then averaged for the zone. This would reduce any bias caused by potential correlation between the two factors. However, the disaggregation of the two factors allows a higher flexibility for inclusion or omission. For this case study, the sensitivity of the characterization factors and results to this aggregation choice was very low (< 1% change).

2.3.4 Limitations

Several remaining limitations in the proposed approach must be mentioned: (1) the effect of temporal variability or seasonality in water use and exposure of aquatic organisms was neglected (only annual averages were used here). It is unclear how seasonality would influence the impacts (higher seasonal stress may be counterbalanced by recovery of the ecosystem after a temporary disturbance) [101, 102]. (2) The causal link between discharge and species richness is not well supported yet. Field-survey

literature [43, 103] is inconclusive, with direction of response even being inverted in some cases. (3) Species are assumed to be distinct within each longitudinal zone, although some species are able to survive in several such zones (e.g. the European eel (*Anguilla anguilla*) occurs over a broad range of longitudinal zones [104]). Additionally, migratory species may inhabit several zones during their life cycle. Literature shows two schools of thought in this respect, which either accept or reject this zonation assumption [81]. The zonation used here [82] is rather coarse. However, already at this level of detail, some species occur in several zones, leading to a slight error of double-counting. Therefore using more detailed zonation schemes could lead to even less distinct species assemblages in each zone, and thus to higher double-counting of species lost. The proposed method is therefore adequate only for regions which exhibit longitudinal zonation with mainly distinct species. (4) The method only accounts for effects in the zone of withdrawal and strictly downstream, excluding feedback effects in upstream zones (such as loss of connectivity and isolation of upstream populations [105]). (5) The weighting by the rarity factor relies on the use of disparate and partially incomplete data sources (e.g. the global occurrence of some species is better documented than for others), which can introduce a bias in the results. Any characterization factors calculated according to the proposed method should be updated when justified by improvements in data quality and completeness.

2.3.5 Recommendations

In general a tiered approach is recommended: the use of species-discharge relationships fitted for the zone-level, specific to the region concerned, and including a maximum of taxa, is recommended in order to capture significant intra-basin spatial variability, and increase precision and completeness. If fitting such a zone-level species-discharge relationship is not possible, using a watershed-level species-discharge relationship for fish developed for a specific region is recommended, in combination with the zone-level characterization factor method suggested here (location can thus be reflected, despite a larger error due to double-counting in the predicted impacts). The cumulative Weibull function may be preferable for modeling species-discharge relationships at latitudes above 42° and other cases where the species-discharge relationship tends towards a limited maximum number of species. If a regionalized SDR is not available for the region of study, using the non-regionalized SDR available [39] is suggested. Furthermore, if longitudinal zonation is not possible or does not exist for a particular river, a watershed-level species-discharge relationship can be used to estimate the loss of species within the whole watershed, which can then be converted to GSE*y using the average rarity and threat status factors for the whole basin. Comparison of potential biodiversity loss between different ecosystems

(e.g. terrestrial and aquatic) using the proposed method would require a correction for missing taxa (such as the normalization suggest here).

3 WATER TEMPERATURE MODEL

3.1 Introduction

The method proposed in Chapter 2 relies on a relationship based on correlation, not mechanistic causality. Therefore the present chapter develops an alternative approach to assess the potential impacts of river water consumption on aquatic biodiversity, shifting the focus from applicability in LCA to use of a causal approach. This approach analyzes the impact pathway relating changes in river discharge to changes in aquatic biodiversity via changes in water temperature, using a deterministic model. This enables the verification of a mechanistic relationship which could explain the correlation observed in the LCIA approach, and provides a more site-specific approach with an increased spatial resolution.

According to the literature overview available in Chapter 1.1, concerning the impacts of climate change on river temperature regimes, as well as the associated impacts on aquatic ecosystems, identified research gaps include the lack of assessment of effects of climate change on river temperature regimes in Switzerland coupled with scenarios of agricultural water withdrawal, and their effects on aquatic biodiversity.

This chapter therefore addresses the first gap by selecting an appropriate modeling framework for the context of far-future climate scenarios at the catchment scale, able to reflect the effects of water withdrawals and riparian shading. This chapter then presents the operationalization, calibration and validation of the model for the case study catchment used in Chapter 2.

The second gap, concerning the assessment of potential impacts of changes in river thermal regimes on aquatic biodiversity, is addressed by linking the above-mentioned modeling approach with two existing methods [56, 70]. The first provides an effect factor for the Rhine river (of which the case study region is a part) relating changes in ambient water temperature to the potentially disappeared fraction of aquatic species. The second uses a water temperature time series to score the ecological quality of the river, based on its deviation from the favourable and tolerable water temperature pattern for expected indicator fish species (identified based on the river type). The results of the impact assessment are integrated in the regional mitigation option assessment in Chapter 5.4.

3.2 Method

3.2.1 Model selection

A previous assessment [106] of existing stream temperature models (with sufficient documentation and with a preference for operational interfaces) was conducted under the co-supervision of the author of the present thesis, in order to select the best model for the present case. This chapter is partly adapted from this assessment, and was completed with further models.

Many models predicting stream and river water temperature have been developed for various purposes. These can be of several types: empirical (based on relationships derived from historical data), deterministic (based on physical relationships) and stochastic (based on statistical relationships). Both empirical and stochastic are however of limited use for the case of predictions in a far future (such as 2050), since the relationships they rely on may no longer be valid in that future. The following criteria [107] were used to assess some of the most widely used and publicly available deterministic models:

- The relationship between discharge and water temperature is directly modeled
- The model does not require excessive data input for other parameters not relevant for water temperature (many models serve primarily to predict water quality for example)
- The input data required (in particular climatic parameters) are covered by standard future projections of climate (currently measurable climatic parameters, often used in river temperature models, are not always predicted in future climate projections due to excessive uncertainties: typically concerned are wind speed and cloud cover).
- The model can address an entire watershed, not just a single stream reach or series of reaches, with a reasonable computation time and input data requirement (indeed, since the irrigation caused by the regional adaptation strategies is distributed throughout the watershed, it is important to consider also the changes in water temperature in a spatially explicit way).
- The model includes a consideration of uncertainties, which can be expected due to the spatial simplifications induced by the above-mentioned entire watershed coverage with limited data and computational requirements.

The models assessed were SSTemp [62], HeatSource [63], SNTEMP [64], QUAL2KW [65], CE-QUAL-W2 [66], LARSIM-WT [67], the approach by Caissie et al. (2005) [68], the approach by Yearsley (2012) [69], and the iWaQa model (unpublished). SSTemp and HeatSource are adapted for reach-scale temperature predictions, but have an excessive data and stability criteria requirement for an entire watershed study; they also are not adequate for predictions largely outside the calibrated value range.

SNTEMP, QUAL2K, and CE-QUAL-W2 were developed for watershed-scale studies, but require a large amount of input data that was unavailable for this study, and do not include an uncertainty assessment. They model the catchment as a network of reaches, where the temperature in each reach is computed and routed throughout the network. Thus for a watershed of the size of the Broye (with 3666 stream reaches defined by junctions alone), the computational time would be excessive. LARSIM-WT would be a good candidate, as it was specifically developed for larger watersheds and basins and has also been applied in the context of climate change; however, it also uses the reach-based approach, and uses discharge as a dynamic variable. While this would be an advantage if investigating precise questions such as the timing of irrigation, it requires heavy computing effort, and is superfluous for the case of smaller catchments with moderate slopes (such as in the case of the Broye). The approach by Caissie et al. (2005) uses an approach similar to that used in the iWaQa model (relying on equilibrium temperature), however it is highly simplified and relies on only air temperature and water levels as drivers, thus not allowing the assessment of mitigation scenarios such as riparian shading. The approach by Yearsley (2012) is also very similar to the iWaQa model in principle, but was not yet available for the present assessment. It uses a grid-based approach which calculates the heat-balance equation for each cell and routes it downstream: again, this is more computationally intensive than the iWaQa model. The iWaQa model was specifically developed for watershed-scale assessments under far future climate scenarios, in the iWaQa project (Integrated Water Quality) [108], also part of the NRP 61 "Sustainable Water Use". It thus has adequate data requirements compared to the availability in future scenarios. The temperature prediction module can be used independently, avoiding irrelevant input data requirements and speeding up computation time. It also includes an assessment of uncertainties, and uses Bayesian inference to deal with the non-uniqueness of calibrated parameter sets (again caused by the simplifications made for watershed-scale and low-input modeling). This model was therefore retained for this study.

3.2.2 Model description

Figure 9 illustrates the components and data flow of the iWaQa water temperature model. The model details (such as the governing equations and principles, and technical requirements) are provided in Annex A2.

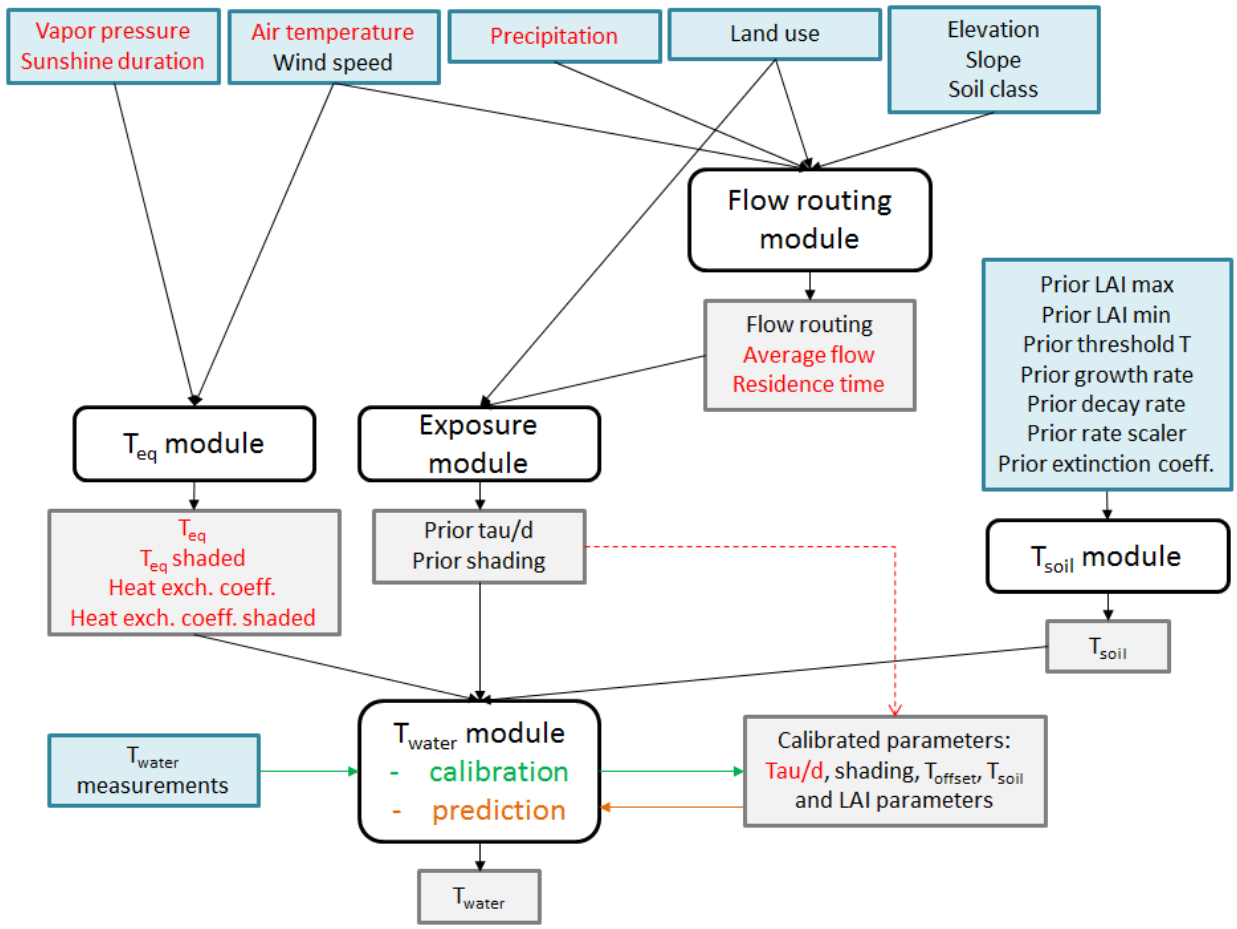


Figure 9: *structure of the iWaQa water temperature model. Rounded boxes represent the different modules. Blue boxes are the required input data and assumptions. Gray boxes are the calculated outputs from each module. Elements which change for future climate are highlighted in red. Elements which are relative for calibration only are highlighted in green, whereas elements used for prediction only are highlighted in orange.*

3.2.3 Model implementation and data sources

The work of Gorski (2012) [106] (co-supervised by the author of this thesis) is acknowledged for the collection and pre-processing of a large part of the input data required by the iWaQa model for the case study used in this thesis. The details of the input data used to implement the model for the case study are provided in Annex A2 (including all the parameter values, distributions, and assumptions for constants used in the model). The principal data sources are listed in Table 5.

Table 5: *data sources for the temperature model.*

Data	Source
Land use	Swiss Federal Office for Statistics [109]

Soil class	Swiss Federal Office for Statistics [110]
Climatological time series	Meteoswiss [111]
Digital elevation model and slope	Swisstopo [96]
Current irrigation withdrawals	Robra and Mastrullo 2011 [112]

3.2.4 Model calibration

In this study, the focus was on the water temperature in the lower part of the Broye case-study watershed of this thesis (see Chapter 4.1.2 for description), where most irrigation withdrawals are concentrated and where temperatures are highest, respectively shading is most limited. 20 measurement stations were installed in the field, in order to provide the time series of observed water temperature required for calibration (their location within the catchment is shown in Figure 10).

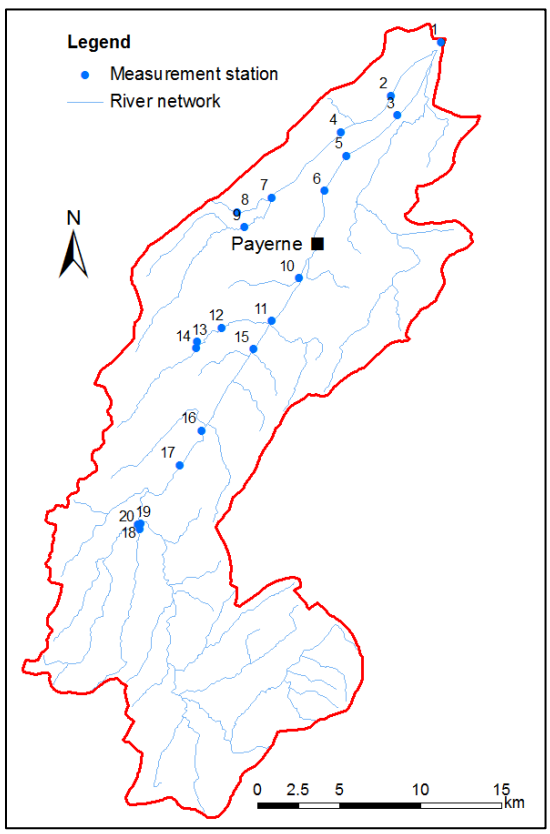


Figure 10: *location of measurement stations in the Broye watershed.*

The stations were installed at the outlets of a selection of sub-catchments within the watershed, representative of the range of hydrological response units occurring within the watershed, and of

stream orders. Furthermore, the locations had to be accessible and beyond a minimal distance from intersections and perturbing factors (e.g. waste-water treatment plant outflows). Hydrological response units are defined by a classification of the whole watershed into units which have homogeneous rainfall-runoff characteristics (based on land use, saturated hydraulic conductivity of the soil, and elevation). This simplifies the spatial variability in the basin, while accounting for the differences in the most important factors for the runoff estimation. The classification is given in Table A2.4. Details of the data loggers used are provided in Annex A2.7.

The measurement period focused on the summer season where flows are lowest and temperatures are highest, and thus extended from April 2012 to October 2012. The calibration period consists of a random selection for each station of two-week blocks, forming in total 50% of the total measurement period (the rest being used for model validation). The calibration was performed for the lower part of the catchment, separately for the main channel measurement stations and the tributary stream measurement stations (indeed, these two groups showed a different behaviour; calibration for both at the same time did not yield satisfactory results). The actual calibration method was an integrated part of the iWaQa model, and is therefore also described in Annex A2.2.

3.2.5 Model validation

The model performance is assessed using the Nash-Sutcliffe coefficient (NSC) and the root mean square error (RMSE) [68]. The NSC assesses to which extent the variability in the observations is reflected in the simulations, thus quantifying the predictive power of the model: it varies from $-\infty$ to 1, 1 being a perfect model performance (where the model predictions perfectly match the observed data), and 0 representing the case where the observed mean predicts the observations just as well as the model does (in other words, the closer to 1, the better the model performance). The RMSE assesses whether the predicted values are close to the observed values (the smaller the RMSE, the better the model performance). These indicators of model performance were calculated using the measurement data that was not used for calibration (50% of total measurements).

3.2.6 Riparian shading scenario

The objective here was to assess the effects on water temperature of climate change alone, of irrigation under the future climate, and of increased riparian vegetation as a mitigation measure. The reference scenario uses a 20 year simulation of climatic data representative of the current climate, whereas the future scenarios use a 20 year simulation of climatic data representative for the year 2050. The assessment focused on a "worst case" perspective, thus the climate change scenario considered

corresponds to the scenario "Ext2050" used in the farm and regional scenario LCAs, and the irrigation scenario considered corresponds to the scenario "productivity" from the regional scenarios (see Chapter 5.1.3). Irrigation withdrawals averaged over the half years were modeled as a decrease in the seasonal river flow, routed from the location of extraction to the outlet of the watershed. In practice in the model, this leads to an increase of the tau/d of the affected stream and river reaches. The extraction locations were attributed according to proximity of irrigated perimeters, flow availability in the streams and existing withdrawal locations (where infrastructure may already exist).

Riparian shading has been shown to be a possible mitigation option to reduce summer water temperatures [113, 114]. The riparian shading scenario here assumes that the entire stream network has riparian vegetation that provides a shading factor of 0.5. This is a realistic factor for streams of the width of the Broye channel (approx. 12 m), whose banks would be fully planted with trees of riparian species (approx. 20m in height) [115]. This is implemented in the model by using a shading factor of 0.5 for all stations rather than the calibrated shading factor.

3.3 Results and discussion

3.3.1 Calibration and validation

The calibrated parameters are provided in Tables A2.9 and A2.10 in the Annex. The measurement period extended from 20.04.2012 to 20.10.2012, covering the 6 months of the summer half year, which is the period of interest in this study (when high temperatures and irrigation occur). Thus calibration is principally valid only for this period of the year. The calibration period used (50% of the measurement period) is rather short: ideally, calibration periods should cover several years. Thus inter-annual variability could not be reflected in the calibration.

Table 6 provides the RMSE and NSC for all stations for the validation period.

Table 6: *indicators of model fit per measurement station.*

Station number	Classification	Root mean square error	Nash-Sutcliffe coefficient
1	Main channel	1.66	0.75
2	Tributary stream	1.10	0.81
3	Main channel	1.44	0.74
4	Tributary stream	1.13	0.79

5	Main channel	1.66	0.78
6	Main channel	1.55	0.86
7	Tributary stream	0.70	0.88
8	Tributary stream	0.66	0.88
9	Tributary stream	0.81	0.83
10	Main channel	1.14	0.86
11	Main channel	1.42	0.80
12	Tributary stream	0.69	0.87
13	Tributary stream	0.75	0.82
14	Tributary stream	0.77	0.83
15	Main channel	1.25	0.84
16	Main channel	0.96	0.92
17	Main channel	0.96	0.83
18	Tributary stream	1.70	0.72
19	Tributary stream	0.84	0.86
20	Tributary stream	1.02	0.82

The root mean square error ranges from 0.66 to 1.70: this is similar to what is achieved by other temperature models [52, 68]. The NSC ranges from 0.72 to 0.92, which is slightly lower than other models have achieved [68], but nevertheless considered sufficient here (an NSC above 0 indicates satisfactory performance; above 0.7, the performance can be considered as good). This lower NSC can be explained by the short measurement period, resulting in lower variability in the observed data. This leads to a lower NSC despite equivalent model performance.

As a further visual assessment, Figure 11 provides typical examples of the simulated and measured temperature time series during the calibration and validation periods, for a channel station and a stream station.

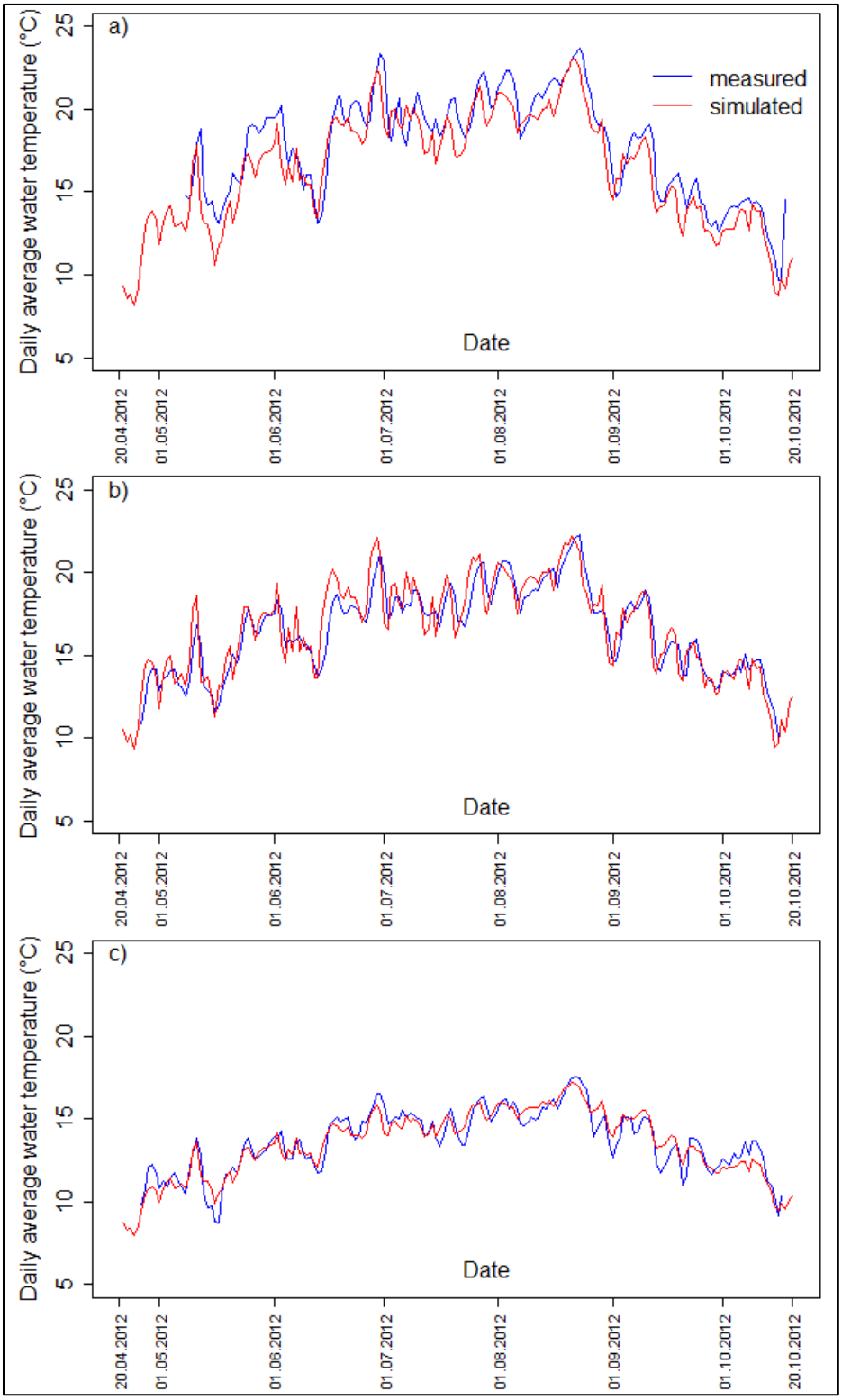


Figure 11: *examples of measured (blue) and simulated (red) temperature time series for measurement period; a) typical channel station (station 1 at mouth); b) typical large tributary station (station 2); c) typical small tributary station (station 12).*

The calibrated model tends to underestimate temperatures during the entire period, at all channel stations, by 0.36 °C to 1.08 °C on average. This is roughly in the order of magnitude of uncertainties related to the data measurements and is therefore considered acceptable. A possible explanation for this bias resides in the simplicity of the soil temperature model and in particular the estimation of LAI decay, which have been found in previous case studies to not capture all characteristics in a satisfactory manner [116]. Station 18 is the only station for which the model has an unsatisfying performance, based on the visual verification (see Figure A2.1). This is confirmed by the RMSE and NSC, for which station 18 performs worst. An explanation for the poor performance at this station is the particularly short measurement time for this station (due to logger losses). Large tributary stations generally perform well, and small tributary stations have a good average performance but overestimate some of the peak low temperatures.

3.3.2 Model limitations

The influence of groundwater infiltration and of heat inputs such as wastewater treatment plants are not considered explicitly in this study. However such effects can be accounted for implicitly during the calibration, via the temperature correction parameter "Toffset" (see model description in Annex A2). Irrigation effects are only modeled for half-years. Thus detailed time-planning of irrigation (e.g. stepwise withdrawals, during the night etc.) cannot be assessed. The perspective assessed here is rather that of the effects of long-term average changes rather than peak situations.

4 LIFE CYCLE ASSESSMENT OF FARM ADAPTATION SCENARIOS

This chapter is structured according to the ISO 14040:2006 standard for LCA [117], with four main sections: goal and scope, life cycle inventory, life cycle impact assessment, and interpretation containing a discussion section.

4.1 Goal and scope

4.1.1 Goals

This chapter presents the life cycle assessment of a selection of farm adaptation scenarios developed in the AGWAM project [7]. These scenarios optimize farm management in order to maximize the certainty equivalent of the farm (a combined indicator of profit maximization and minimization of profit variability). For a description of the farm optimization model that served to produce these scenarios, see Annex A3.1 and Lehmann (2013) [7].

The objectives of this life cycle assessment are to:

- Evaluate the environmental impacts of farm adaptation scenarios to climate change for two case study regions in Switzerland.
- Identify an optimal environmental impact indicator set in this context.
- Identify trade-offs between environmental indicators, and with the economic objectives of the adaptation scenarios.
- Determine whether the adaptation scenarios also mitigate climate change (by reductions in their global warming potential).
- Provide an application example for the LCIA method developed in Chapter 2, in the context of a full LCA.

4.1.2 Case study regions

The case study regions used in this chapter are illustrated in Figure 12:

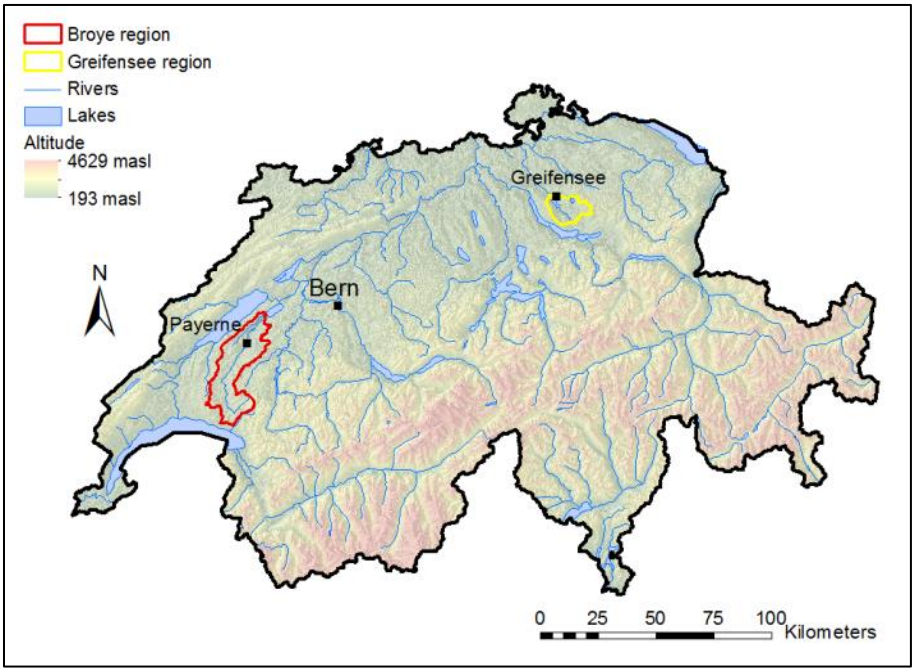


Figure 12: *case study regions Broye and Greifensee.*

The Broye region consists of a watershed of 598 km^2. Its principal river is the Broye (average discharge 11.73 m^3/s at the outlet into lake Morat [118]). The region can be divided into a hill area (max. altitude 1'500 masl, average temperature 7.1 °C and average precipitation 1'535 mm/y at weather station Semsales [119]), and a flatter lowland area (altitude from 400 to 600 masl, average temperature 9.6 °C and average precipitation 886 mm/y at weather station Payerne). Land use is dominantly agricultural, with mixed dairy and arable crop production in the hill area, and dominantly arable crop production in the lowland area [7]. The latter is also an important potato production region in Switzerland. Significant amounts of irrigation water, mainly sourced from the river Broye, are already required in the present climate [112]. This water is currently price-free (and capital costs such as irrigation infrastructure tend to be heavily subsidized). Climate change is expected to affect the lowland area more severely, therefore the farm scale models focused on this part of the Broye region.

The Greifensee region is a watershed of 164 km^2. Its principal river is the Glatt (average discharge 4.01 m^3/s [118] at the outlet from the Greifensee lake, volume 0.148 km^3 [120]). This region also presents a hill area (maximum altitude 1030 masl, average precipitation 1'388 mm/y and average temperature 9.8 °C at weather station Hinwil) and a lowland region (altitude from 400 to 500 masl, average precipitation 1'187 mm/y and average temperature 10.5 °C [119]). Similarly to the Broye region, arable crop production is concentrated in the lowland region, and dairy production is concentrated in the hill region.

Precipitation in the region is above the optimum level for agriculture, and therefore irrigation is practically only required for vegetable production. For this region too, farm scale models focused on the lowland area.

Both regions show a low water stress according to the method of Pfister et al. (2009) [35], a recent method often used to estimate the impacts of water consumption in LCA, for which the spatial resolution was at the river basin or national level at the time of this thesis.

4.1.3 Scenarios and temporal system boundaries

The scenarios considered in the farm scale assessment are summarized in Table 7 and explained in the following paragraph. A more detailed explanation of the scenario assumptions as applied in the farm optimization model is available in Lehmann (2013) [7]. All scenarios present a modeled farm over one average year of function.

Table 7: *summary of scenarios considered at the farm scale. "Optimum" designates the economically optimized farm (maximizing the certainty equivalent, see description of farm optimization model in Annex A3.1).*

Scenario name	Assumption
Reference	Optimum under current climate, prices and subsidies.
Ext2050 without adaptation	Same crops and animals as in reference, but under extreme climate scenario in 2050
Ext 2050	Optimum under extreme climate scenario in 2050
Mod 2050	Optimum under moderate climate scenario in 2050
Ext 2050, subsidy change	Optimum under extreme climate scenario in 2050, with subsidy change
Ext 2050, European prices	Optimum under extreme climate scenario in 2050, with product price change
Ext 2050, water pricing	Optimum under extreme climate scenario in 2050, with water pricing
Ext 2050, water quota	Optimum under extreme climate scenario in 2050, with water quota

The reference is the optimized farm under the current climate (i.e. a 25-year simulated daily weather series). This reference does not exactly correspond to the current real situation, since it is the result of a modeled optimally managed farm. However it is considered realistic [7] and enables an objective comparison of the effect of the scenarios, without including a bias due to the effect of the model compared to reality.

The climate change scenarios considered reflect the climate projected for 2050 in the case study regions, based on the A1B IPCC emissions scenario [23]. Both an extreme and a moderate climate change signal were available, developed using the ETH CLM regional model and the SMHIRCA regional model respectively, as provided in the ENSEMBLES project [121]. Climate was downscaled to a 25-year series of daily weather data required for the crop growth model using the weather generator LARS WG [119]. In this thesis, these scenarios are called "Ext2050" and "Mod2050" respectively.

Additionally, the situation under the future climate if the farm does not adapt was assessed: in this case, farm management is the same as in the reference (no adaptation), but climate data corresponds to the Ext2050 scenario. In detail, the quantity of water for irrigation changes slightly, because in the farm optimization model, the fixed management variable is not the quantity of water, but the soil water depletion threshold that triggers irrigation in the model. Thus if the climate is hotter and dryer, more irrigation water will be applied in the model, even if the management (i.e. soil water depletion threshold) stays the same (i.e. is not adapted to the future climate). Similarly, fertilization amount varies slightly, in order to respect legislation which regulates the application of fertilizer in relation to the yield (and the latter changes due to the change of climate; see Lehmann (2013) [7] for further explanation of the assumptions in this scenario). This scenario is called "Ext2050 without adaptation" here.

This selection of scenarios allows the assessment of the impacts of climate change in a worst-case situation (extreme climate change), as well as the sensitivity of impacts to the choice of climate change scenario (by including a second, more moderate scenario). The assessment of the effect of economic optimization under future climate is also possible by the inclusion of scenarios both with and without adaptation.

The two policy scenarios considered [7] consist of 1) changes in direct subsidies according to currently planned changes in the Swiss subsidy system. These changes would principally reduce subsidizing (see detail in Table A3.2). This scenario is called "subsidy change"; 2) changes in agricultural product prices

due to a potential market liberalization with Europe. This would principally cause a decrease in product prices (see detail in Table A3.1). In this scenario, prices were therefore assumed to be at the current European level (instead of the current Swiss level). This scenario is called "European prices". Consideration of these scenarios allowed the assessment of impact sensitivity to changes in prices and subsidies under future climate.

Two water use restriction scenarios were available [7]: 1) water is priced at 1 CHF/m^3; 2) a water quota is fixed at 4000 m^3/y for the farm. These scenarios are called "water pricing" and "water quota" respectively. Such policy measures are considered to have a high potential for restricting the impacts of water use on aquatic ecosystems [47]. Consideration of these scenarios allowed the assessment of impact sensitivity to two possible regulation interventions of water use.

Other elements of influence for which no change was modeled in the farm scenarios in 2050 include prices of inputs (e.g. fuels), and evolution of plant varieties and their characteristics.

4.1.4 Spatial and physical system boundaries

The system boundary considered for the farm LCA was "cradle to gate", which includes all inputs to the farm and their own life cycle impacts, as well as on-farm processes and direct emissions (such as field operations, pesticide, nutrient and greenhouse gas emissions etc.). The following input groups were considered, based on the SALCA methodology for farm LCA [12, 19]:

- Infrastructure and machinery (e.g. stables, storage buildings, tractors, irrigation infrastructure etc.)
- Energy carriers (e.g. diesel, electricity etc.)
- Mineral fertilizers (mineral nitrogen, phosphate, potassium etc.).
- Pesticides
- Seeds
- Water for irrigation (assumed to be river water) and animals (assumed to be tap water).
- Animals for herd replenishment
- Fodder (e.g. concentrated feeds, as well as silo maize and hay in case of insufficient on-farm production)

On-farm processes included field operations, grain and hay drying, storage, milking (in case of dairy cows); manure and slurry were applied only if sufficient on-farm quantities were available as well as sufficient demand for application was present (i.e. first fertilization of arable crops, fertilization of

grasslands). Processes occurring after the farm production system were not considered (e.g. transport of products, processing, retailing).

4.1.5 Functional units

Three functional units (FU) were initially used, representing the three main functions of agriculture [19]: (1) gross profit margin in Swiss francs (CHF), representing the function of generating economic livelihood; (2) megajoules digestible energy for humans (MJ dig. en.) produced, representing the function of food production. This FU enables aggregation of different products into a single unit (as opposed to the FU "kg", commonly used to represent the production function for single-product LCAs); (3) ha*y, representing the function of land management and occupation, in the view of optimizing the management of a certain area. However, results per ha*y do not consider the productivity achieved by the scenario, and therefore ignore consequential impacts caused by a potential decrease in productivity. The FU MJ dig. en. on the other hand reflects the double goal of minimizing impacts per area while maximizing productivity per area [122]. It has been argued that both the production FU and the land occupation FU should be used in parallel, since the land occupation can give indications of impacts which may accumulate in a region and cause severe regional problems [10]. This is relevant for the regional scale assessment, where regional problems are addressed, and is therefore not required at the farm scale in this particular study. Additionally, the FU ha*y did not vary between the scenarios modeled here, therefore this does not bring additional information in the present study. Crop yields and livestock production were converted to MJ dig. en. following a previous method for Swiss agricultural LCA [22], using the average MJ content available to human digestion, per kg product. Animal production (meat, milk) is likewise converted to MJ available to human digestion per kg product, although animal products may exhibit a lower calorie density while providing more proteins for example (as discussed in Chapter 4.4.10). Therefore the results for purely crop farms are distinguished from the results for mixed farms (with both crop and animal production), which makes farms more comparable.

4.1.6 Method selection

There is less experience of application of LCA to whole farm systems compared to that of individual products, and therefore the choice of method is less clearly defined by any consensus. Recommendations for indicator-based methods assessing environmental impacts at the farm scale include: 1) considering a range of both local and global indicators; 2) prioritizing indicators of environmental impacts rather than indicators based on changes in practice (which do not model the subsequent impact); and 3) using units of quantifiable impacts (e.g. amount of greenhouse gas equivalents released) rather than unit-less scores as the indicator output [10]. The only existing

operational methodology for farm LCA in Switzerland, including inventory and impact assessment tools as well as a database of flows relevant for agricultural production (e.g. agricultural machinery, infrastructure, processes, crops) is SALCA [12]. Further impact assessment approaches developed for farms include REPRO [13] and RISE [14] and the approach of Eckert et al. (2000) [15]. REPRO models the energy and mass cycling in the farm and evaluates several environmental indicators such as energy balance, GHG emissions, carbon fluxes, soil compaction, erosion risk and on-site biodiversity. However this does not include the entire life cycle of the farm, and omits certain environmental impact categories such as eutrophication, toxicity and acidification. RISE assesses farm sustainability with environmental, economic and social criteria, however the environmental criteria do not model impact pathways all the way to actual impacts, and are also less complete than in SALCA (again, eutrophication, toxicity and acidification in particular are missing). Similarly, the approach of Eckert et al. (2000) covers less impact categories and does not model some indicators all the way to an impact. According to a comparative study of impact assessment approaches for agricultural systems [123], SALCA had a high to highest performance in scientific soundness, including coverage of environmental issues, production branches and production factors, depth of environmental analysis, avoidance of incorrect conclusions, and transparency. The ranking of impacts in a comparative assessment of systems using SALCA or REPRO was correlated, therefore a significant comparative error caused by the choice of one or the other method is not expected. The SALCA methodology was therefore selected for this thesis: additional advantages are that it has been specifically developed for the Swiss context, has been widely applied to farm systems, and certain calculations can be parameterized in order to adapt the tools to future climatic conditions for example.

4.1.7 Environmental indicator selection

Table 8 lists the initial set of 13 environmental indicators used for assessment, based on their relevance for farm systems [19], expected relevance in the study context (i.e. aquatic biodiversity) and interest expressed by stakeholders (i.e. terrestrial biodiversity).

Table 8: *initial indicators considered.*

Indicator (impact category)	Method	Description	Important contributors in agriculture
Non-renewable	ecoinvent [124]	Direct and indirect energy	Diesel, fertilizer

energy demand		resource use (including fossil resources and uranium)	production
Global warming potential (GWP) for 100 years	IPCC [23]	Emissions of carbon dioxide, dinitrogen oxide, methane	Fertilizer production, organic fertilizer application, animals
Tropospheric ozone formation potential	EDIP 2003 [125]	Emissions of nitrogen oxides, volatile organic compounds	Exhaust gases
Acidification potential	EDIP 2003 [125]	With and without regionalization	
Freshwater eutrophication, Marine eutrophication	ReCiPe [26], EDIP 2003 [125]	Nutrient enrichment of ecosystems due to emissions of nitrogen and phosphorous	Fertilizer application
Terrestrial eutrophication	EDIP 2003 [125]		
Aquatic ecotoxicity potential, Terrestrial ecotoxicity potential (TEP), Human toxicity potential	UseTox (using UseTox-recommended factors only) [126], CML [127]	Toxic emissions to ecosystems and humans	Pesticide application, heavy metal emissions from fertilizers
Land use competition	ecoinvent [124]	Total land area and time occupied	Direct land occupation, fodder imports
Potential aquatic biodiversity loss (ABL)	Chapter 2, Tendall et al. 2013 [128]	Fish species richness loss due to on-site river water consumption	On-site irrigation
Reduction of potential terrestrial biodiversity	SALCA Biodiversity (adapted) [129]	Decrease in on-site potential terrestrial biodiversity relative to the	Crop mix, fertilization intensity, stocking

	best score achievable	density

The initial impact category set (see Table 8) was calculated for all scenarios at the farm scale. However, the LCA approach used here provides results for too many environmental categories in order to be recommended for stakeholder communication [130], and contains a certain amount of redundancy in the information [131]. Therefore a selection of indicators was performed, identifying the impact categories which optimize the information content (based on the initial indicators used, listed in Table 8) for the farm and regional scale in a climate change context. This selection was done using principal component analysis (PCA), as previously applied in the LCA of farming systems [20] and of crops [132]. PCA enables a reduction of the complexity and number of variables (here: indicators) required to explain differences between observations (here: scenarios). This simplifies the analysis and communication of results with stakeholders, by reducing redundancy in information whilst maintaining the major trade-offs between indicators. Statistical modeling was done using the R software [83]; LCA calculation was implemented in the SimaPro software [133]. The reference aquatic biodiversity and terrestrial biodiversity components and their reactions to stress may change by 2050: however this was not considered, due to extremely high uncertainties and difficulty in prediction.

4.2 Life cycle inventory

4.2.1 Inventory modeling

The inventory of all inputs and emissions was established for each farm adaptation scenario, reflecting the changes in the variables and outcomes of these scenarios. The outcomes of the farm optimization model [7] (described in Annex A3.1) in each scenario provided the following inventory flows as required in SALCA: crop land allocation, yields, animal types and numbers, animal water use, nitrogen fertilizer application (quantity and dates), total irrigation water use, sowing dates, harvest dates, organic fertilizer produced, fodder uptake on pasture, total fodder requirement, and soil parameters. For inventory flows which were not provided by the farm optimization model, consistent assumptions were made based on inventories of representative model farms of the same type (previously developed for Switzerland [22]), for farm infrastructure, seeds, energy use, organic fertilizer application and methods, pesticide application, fertilizer contents and brands; based on reference norms for Swiss agriculture [134] for phosphorous, potassium, calcium and magnesium fertilizer application; based on the AGRAMMON study [135] for animal pasture timing and duration; and based on a study on irrigation efficiency [136] for irrigation infrastructure. Direct farm emissions were modeled using the SALCA

models [137-139]. Background inventory data was sourced from the SALCA database [140] and from the ecoinvent database [141]. The inventories for all farm scenarios are provided in Annex A3.8, Tables A3.7 to A3.10.

4.2.2 Assumptions for 2050

In general, the models and data used for agricultural LCA are valid for the current climate, and could not be adapted to uncertain situations in 2050 within the scope of this thesis; however, several exceptions were possible for important components of agricultural LCA: erosion and nitrate leaching were taken from the outcomes of the crop growth model used in the farm optimization module (which are valid for the climate in 2050) instead of using the SALCA direct emission calculation methods for these two parameters (valid only for the current climate). Pesticide application has a large influence on the toxicity impacts of a farm, and can be expected to change by 2050, although it is hard to quantify exactly how. Therefore an expert workshop was conducted to discuss evolution trends of pesticide application, the outcomes of which (presented in Chapter 4.4.7) provided a qualitative adjustment of the expected impacts related to pesticide use (also presented in Chapter 4.4.7).

4.3 Life cycle impact assessment

This chapter presents the results of the life cycle impact assessment. The interpretation of these results and their causes is provided in Chapter 4.4 "Interpretation", following to the ISO 14040:2006 standard for LCA [117].

4.3.1 Selected indicators

Eutrophication was assessed using both CML and ReCiPe (as shown in Table 8). The results for both these methods were found to be highly correlated in this study (R^2>0.9). ReCiPe is a more recent method and was therefore retained as calculation method for this impact category. Ecotoxicity and human toxicity were assessed using both CML adapted to include additional pesticides relevant for agriculture [142] and UseTox (recommended factor set proposed by the UseTox method) (see Table 8). Correlation was lower (0.5<R^2<1). Several major pesticides relevant for the current study are not considered in UseTox. Therefore despite this method being more recent, the adapted CML method was retained as more relevant for this case.

The principal components and loadings of the PCA after varimax rotation are shown in Table 9. Five components are retained using a parallel analysis as well as an optimal coordinates analysis. Three

components are retained based on the acceleration factor. Only two components are retained using a threshold eigenvalue of 1.

Table 9: *principal component analysis (PCA): rotated component matrix with twelve impact categories (per MJ dig. en.), for N=22 scenarios (both regions, both climate scenarios, and policy scenarios as listed in Table 7). Loadings above 0.5 (respectively below -0.5) are in bold. CML+ designates the CML method extended to additional relevant pesticides.*

	Component				
	1	2	3	4	5
Impact categories					
Non-renewable energy demand	-0.30	0.08	0.06	0.15	0.33
Global warming potential	-0.30	-0.04	-0.13	-0.02	0.16
Tropospheric ozone formation potential	-0.30	0.02	-0.07	0.03	0.15
Acidification potential	-0.29	-0.09	-0.21	-0.09	0.10
Terrestrial eutrophication (ReCiPe)	-0.29	-0.10	-0.23	-0.11	0.09
Marine eutrophication (ReCiPe)	-0.28	-0.07	-0.15	0.53	**-0.73**
Freshwater eutrophication (ReCiPe)	-0.30	-0.05	-0.07	0.19	-0.05
Human toxicity potential (CML+)	-0.30	0.08	0.05	0.13	0.24
Terrestrial ecotoxicity potential (CML+)	-0.20	**0.58**	**0.69**	0.03	-0.13
Aquatic ecotoxicity potential (CML+)	-0.29	0.12	-0.00	0.21	0.27
Land use competition	-0.29	0.00	-0.05	-0.44	-0.21
Potential aquatic biodiversity loss	0.14	**0.78**	**-0.61**	-0.03	-0.03
Reduction of potential terrestrial biodiversity	-0.28	0.03	0.04	**-0.62**	-0.32
Total variance explained					
Initial eigenvalues	11.23	1.03	0.44	0.22	0.06
Variance explained (% of variance)	84.73	8.98	3.05	2.53	0.55

The loadings in the first component are quite homogeneously distributed (except for potential aquatic biodiversity loss), reflecting the structure of the farm adaptation scenarios: many of the management options that would naturally vary independently are not explicit variables in these scenarios (e.g. pesticide application [21]: this was considered too uncertain to assume changes by 2050 in the economic farm model [7] and is therefore not a variable; the possible evolution of pesticide application

is discussed in Chapter 4.4.7). Potential aquatic biodiversity loss shows an inverse loading of the first component compared to all the other impact categories; in addition, the second component is dominated by impacts on aquatic biodiversity. This shows that in the context of climate change (as modeled in this study), it is important to consider impacts of water use as an additional dimension of environmental impacts, even for regions which are not identified as having high water stress using the Water Stress Index [35] for example. The third, fourth and fifth components are dominated by terrestrial ecotoxicity potential, reduction of potential terrestrial biodiversity and marine eutrophication respectively. A correlation analysis (Table A3.4 in the Annex) showed that marine eutrophication and reduction in potential terrestrial biodiversity are highly correlated with the other indicators loading the first component ($R^2>0.9$) whereas terrestrial ecotoxicity has a lower correlation with these indicators ($0.52<R^2<0.72$), and aquatic biodiversity loss is inversely correlated to all other indicators ($-0.03>R^2>-0.49$).

Based on these results, three environmental impact categories are retained in the detailed analysis below: (1) global warming potential (GWP, expressed as kg CO_2 equivalents), considered representative for the indicators loading the first component. Global warming potential has the advantage of being a generally well-known concept (as recommended by Mouron et al. (2006) [20]), as well as providing information on feedback to climate change; (2) potential aquatic biodiversity loss (ABL, expressed as global species extinction equivalents*y); (3) Terrestrial ecotoxicity potential (TEP, expressed as kg 1,4-DB equivalents). Together, these show the principal trade-offs between the scenarios.

4.3.2 Trends in functional units and eco-efficiency

Figure 13, Figure 14 and Figure 15 show respectively the relationships of production (as MJ dig. en. according to the conversion used by Hersener et al. (2011) [22]) to profit (in CHF, as provided by the farm optimization scenarios [7]), global warming potential (as an example of environmental impacts) to production, and global warming potential to profit. This gives an overview of the production, profit and global warming potential for all scenarios, distinguished according to farm type, region and period (current reference or future climate: both "Ext2050" and "Mod2050"). The last two figures allow an analysis of the trend in environmental efficiency according to climate change. A detailed analysis of individual scenarios and causes of trends is available in the next section.

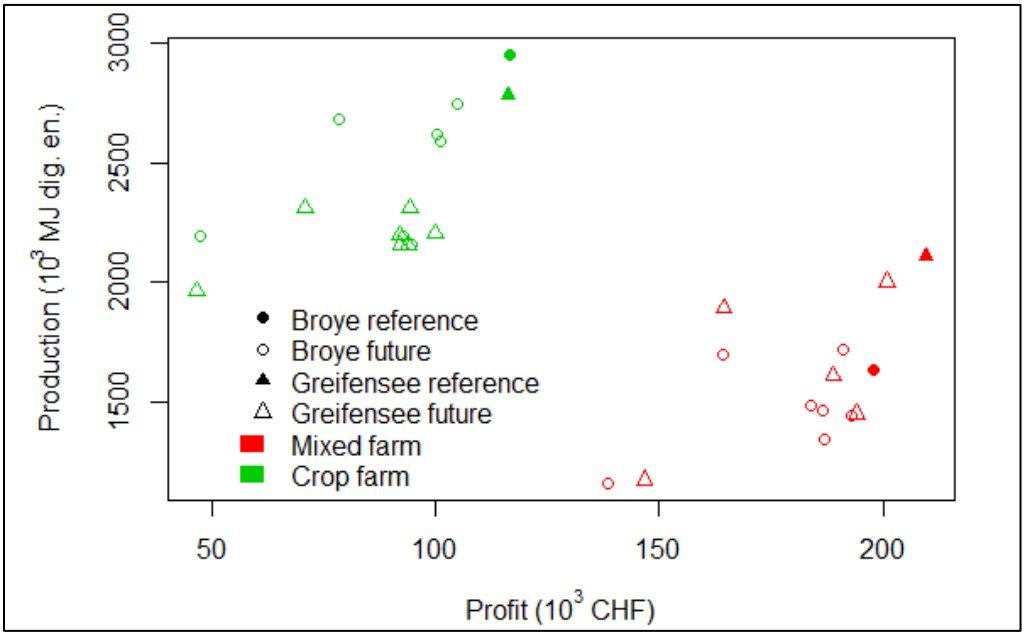


Figure 13: *farm production (as MJ dig. en.) versus farm profit for all scenarios (see Table 7). Colours distinguish farm types (as defined in Annex A3.1), shapes distinguish regions (as presented in Chapter 4.1.2), and symbol fill distinguishes the references from the future scenarios. Mixed farm = farms with arable crops and livestock. Crop farm = farms with arable crops only.*

Figure 13 shows that profit is higher for mixed farms (which combine both arable crops and livestock) for all scenarios, whereas production of energy for humans is higher for crop farms (which produce only arable crops) for almost all scenarios. This is consistent with previous findings for real Swiss farms [22]. Generally crop farms are more productive in the Broye region than in the Greifensee region, and see a decrease in both profit and production in future scenarios. Mixed farms are generally more productive in the Greifensee region than in the Broye region; profit and productivity both decrease in future in the Greifensee region, whereas in the Broye region, a few future scenarios provoke a slight increase in production. Generally an increase in profit is correlated with an increase in production for both farm types.

Global warming efficiency in relation to production is shown in Figure 14 as global warming potential versus the production of energy for humans.

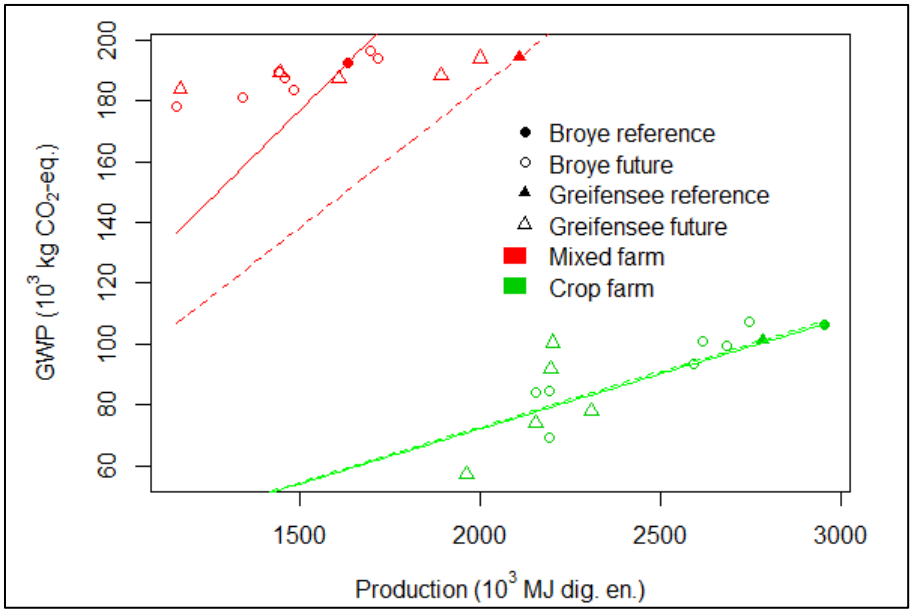


Figure 14: *global warming potential (GWP) versus production (as MJ dig. en.) for all scenarios. Colours distinguish farm types, shapes distinguish regions, and symbol fill distinguishes the references from the future scenarios. As a reference for each farm type, full lines represent the ratio GWP/production for the Broye reference scenario, and dashed lines represent the same ratio for the Greifensee reference scenario.*

An increase in production generally leads to an increase in global warming potential, more markedly for crop farms. Crop farms are more eco-efficient than mixed farms (if considering global warming potential): global warming potential is systematically lower, while achieving similar or higher production. There is no notable difference between the efficiencies of the Broye region and of the Greifensee region for crop farms, whereas for mixed farms, the Greifensee region is more eco-efficient in most cases. Climate change causes a decrease in global warming potential in most cases, but also a decrease in eco-efficiency for mixed farms, and no significant trend for crop farms. The global warming potential is benchmarked against existing literature in Chapter 4.4.6; impacts related to the production functional unit (MJ dig. en.) have been previously shown to be higher for mixed farms than for crop farms in Switzerland, in accordance with the above results [22].

Eco-efficiency is shown in Figure 15 as the global warming potential versus the profit of the farm.

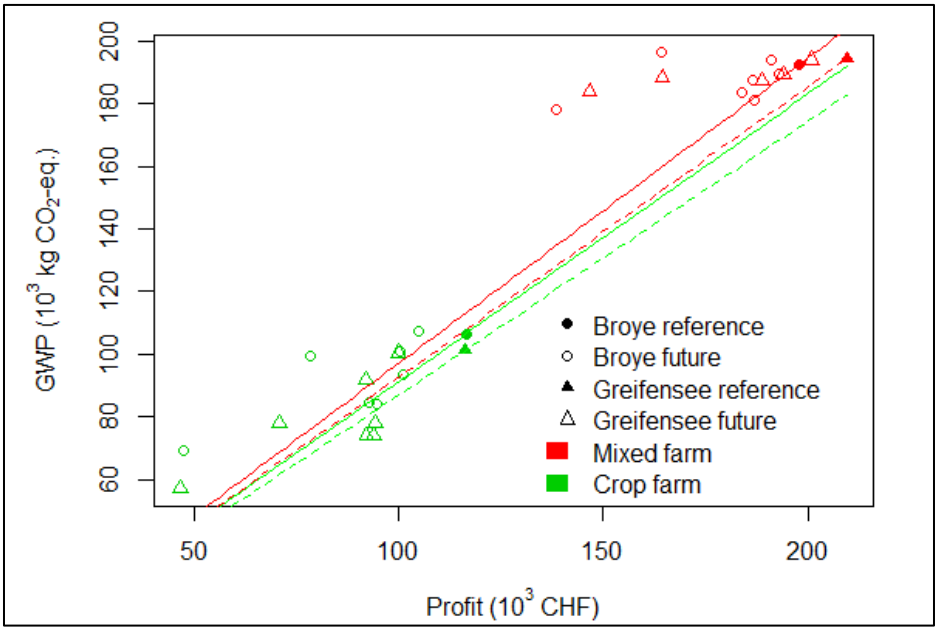


Figure 15: *global warming potential (GWP) versus profit for all scenarios. Colours distinguish farm types, shapes distinguish regions, and symbol fill distinguishes the references from the future scenarios. As a reference for each farm type, full lines represent the ratio GWP/profit for the Broye reference scenario, and dashed lines represent the same ratio for the Greifensee reference scenario.*

Here eco-efficiency (considering global warming potential and economic profit) is similar for both farm types and both regions, whereby mixed farms have a slightly lower eco-efficiency than crop farms, and the Broye region has a slightly lower eco-efficiency than the Greifensee region. Mixed farms have both higher profits and higher global warming potential. In accordance with these results, a previous study of Swiss farms under current conditions likewise showed that global warming potential and profits are higher for mixed farms than arable farms, and that global warming potential per unit profit is slightly higher for mixed farms than for arable farms [22]. Climate change tends to cause a decrease in eco-efficiency for most cases (as shown in Figure 15: most future scenarios are above the current reference lines, meaning that global warming potential per unit profit is higher).

4.3.3 Impacts of climate change and economic optimization

The selected environmental impacts per MJ dig. en. (displayed as relative to the maximum) for optimized farms under current and future climate are shown in Figure 16 for the case of mixed farms, and Figure 17 for the case of crop farms. The future climate is represented by the scenario "Ext2050" with the extreme climate change signal, in order to consider the "worst case". Results are additionally shown for a farm in the future climate, without adaptation to climate change. The results related to CHF and to MJ dig. en. showed very similar trends, therefore only results for MJ dig. en. are shown. The

absolute impacts, as well as the production in MJ dig. en., the profit in CHF and the land occupation in ha*y are shown for all scenarios in Table A3.3 in the Annex. Yields (as MJ dig. en. per ha*y) are additionally summarized in Figure A3.2 in the Annex.

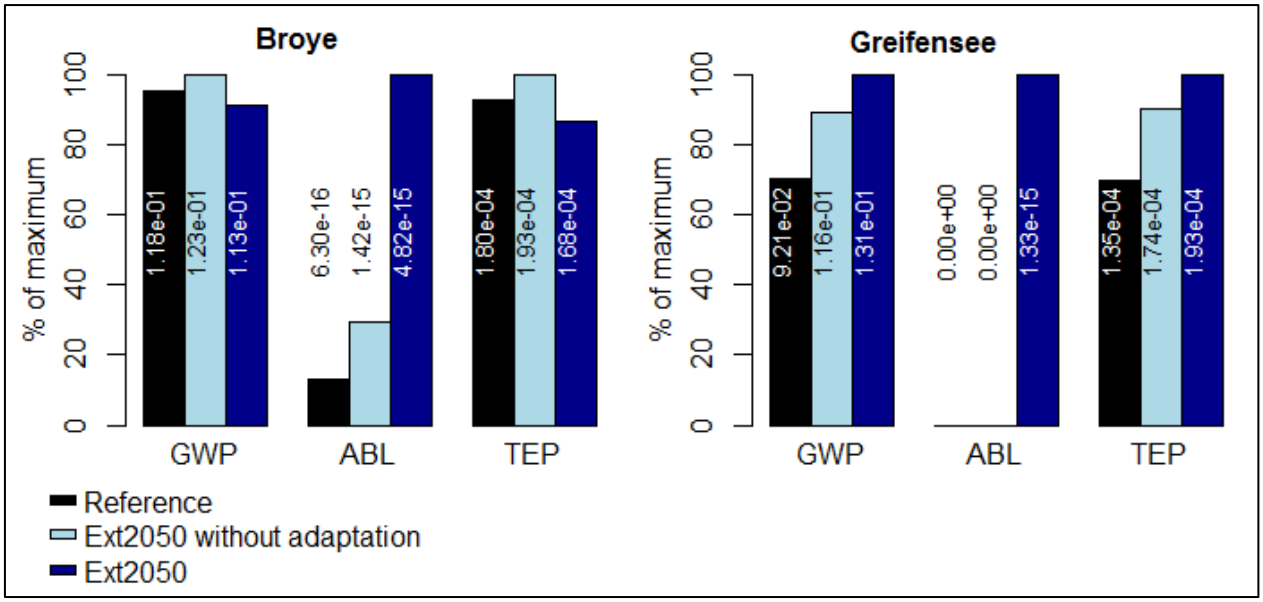


Figure 16: *mixed farm : environmental impacts per MJ dig. energy (relative to the maximum), for regions Broye and Greifensee, under current and future climate. The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

For mixed farms, all impacts per MJ dig. en. increase if no adaptation to climate change occurs (except aquatic biodiversity loss in the Greifensee). Aquatic biodiversity loss increases for optimization under future climate. Global warming potential and terrestrial ecotoxicity potential decrease slightly for optimization under future climate in the Broye, but increase in the Greifensee.

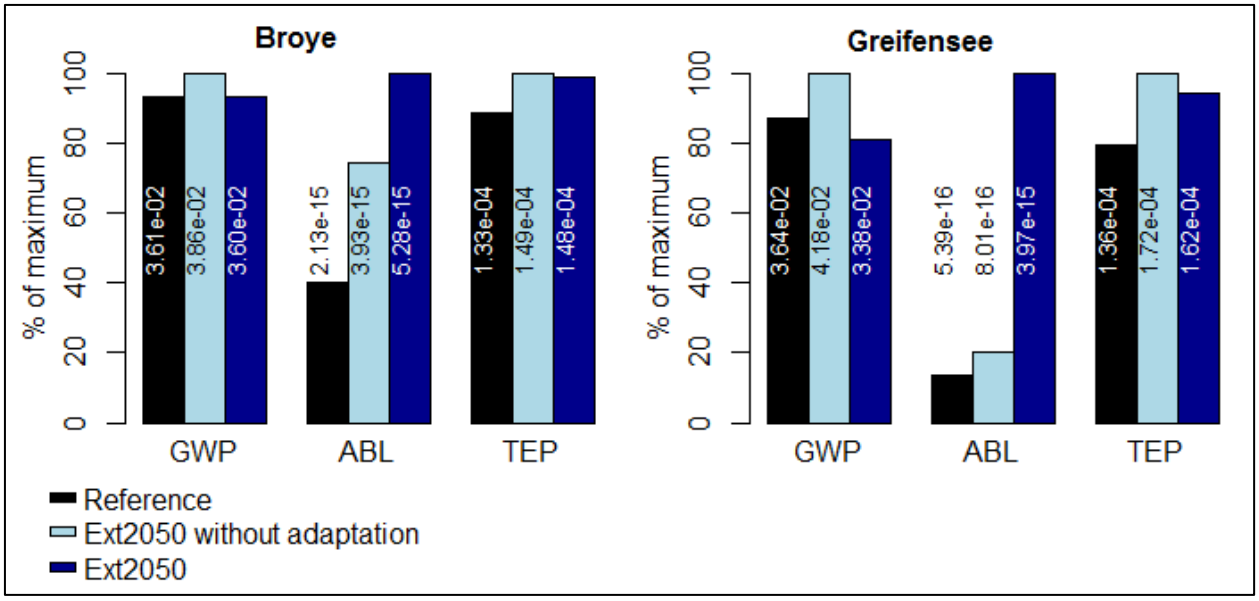


Figure 17: *crop farm : environmental impacts per MJ dig. energy (relative to the maximum), for regions Broye and Greifensee, under current and future climate. The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

For crop farms, all impacts increase if no adaptation to climate change occurs. Aquatic biodiversity loss and terrestrial ecotoxicity potential also increase for optimization under future climate, compared to the current situation; only aquatic biodiversity loss is higher with optimization than without. Global warming potential is stable or decreases for optimization under future climate, compared to the current situation.

4.3.4 Water restriction scenarios

Options to reduce aquatic biodiversity loss caused by optimization include setting a price on water and attributing a water quota to the farm, in order to limit irrigation water consumption. The impacts of optimization under such scenarios are shown in Figure 18, for the example of the Broye region (for both mixed and crop farms).

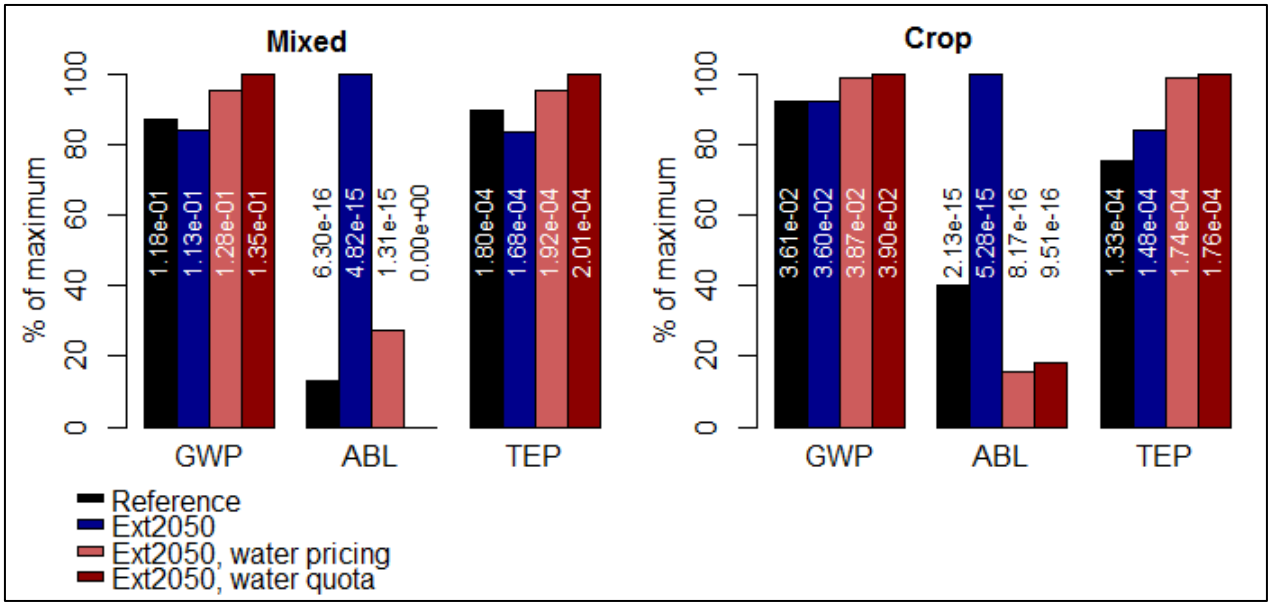


Figure 18: *environmental impacts per MJ dig. en. (relative to the maximum) of optimized farms under future climate and water use restriction scenarios, for the Broye region. The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

Aquatic biodiversity loss is reduced by 75 to 100% with both water use restriction scenarios, for both farm types. Global warming potential and terrestrial ecotoxicity potential increase with both water use restriction scenarios, for both farm types, by 10 to 20%. The relative importance of these changes are discussed in the interpretation section (Chapter 4.4.2).

4.3.5 Sensitivity to climate scenario

The impacts of the optimized farm according to the different future climate scenarios are shown in Figure 19 for mixed farms and Figure 20 for crop farms.

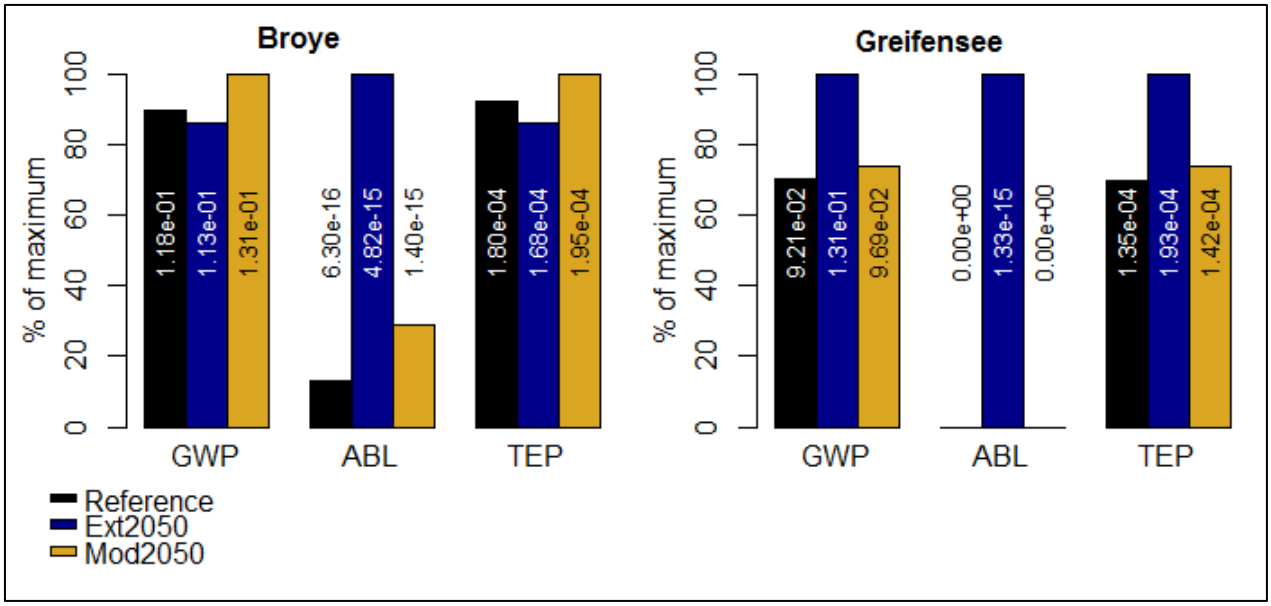


Figure 19: *mixed farm: environmental impacts of optimized farm under different future climate scenarios (extreme change signal and moderate change signal), per MJ dig. en (relative to maximum). The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

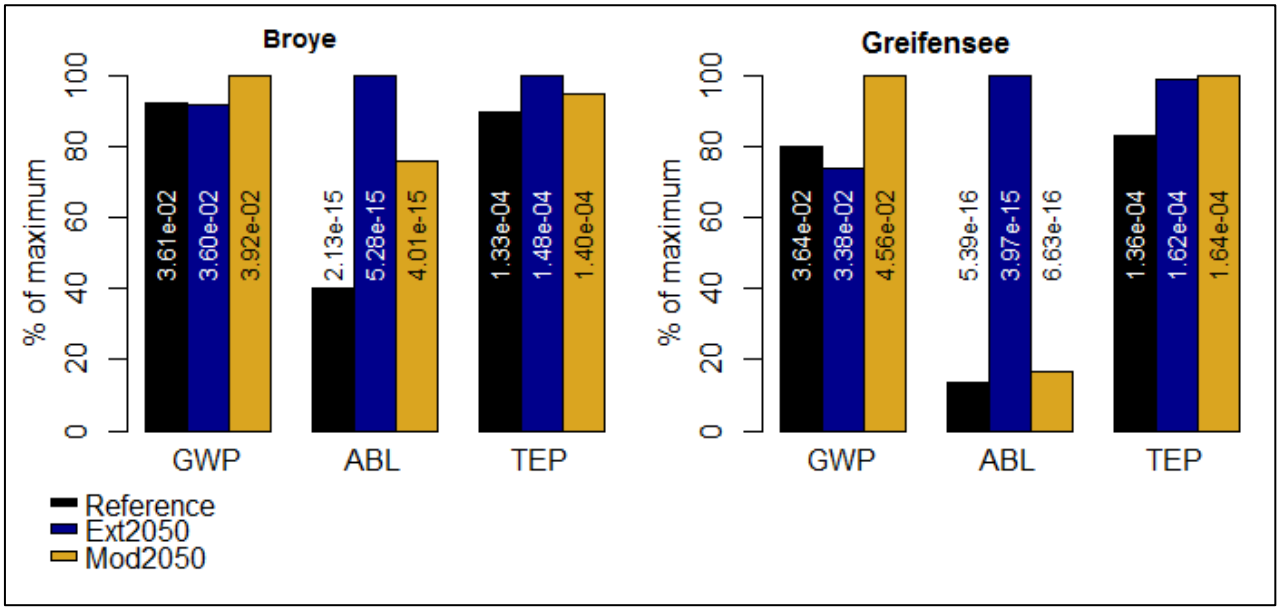


Figure 20: *crop farm: environmental impacts of optimized farm under different future climate scenarios (extreme change signal and moderate change signal), per MJ dig. en (relative to maximum). The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

For both crop farms and mixed farms, aquatic biodiversity loss is highest under extreme climate change. Impacts under moderate climate change increase compared to the reference in all cases, except for aquatic biodiversity loss for mixed farms in the Greifensee region. Compared to extreme climate change, moderate climate change leads to higher impacts in the Broye region (except aquatic biodiversity loss, and terrestrial ecotoxicity potential for the case of crop farms); and in the Greifensee region, lower impacts for mixed farms, but higher impacts for crop farms (again except for aquatic biodiversity loss). The interpretation of these results is available in Chapter 4.4.3.

4.3.6 Sensitivity to price and subsidy scenario

The impacts of the optimized farm according to the different price and subsidy scenarios under future climate are shown in Figure 21 for mixed farms and Figure 22 for crop farms (note that this analysis does not consider policies which affect water pricing and quotas, as these have already been discussed in a previous section).

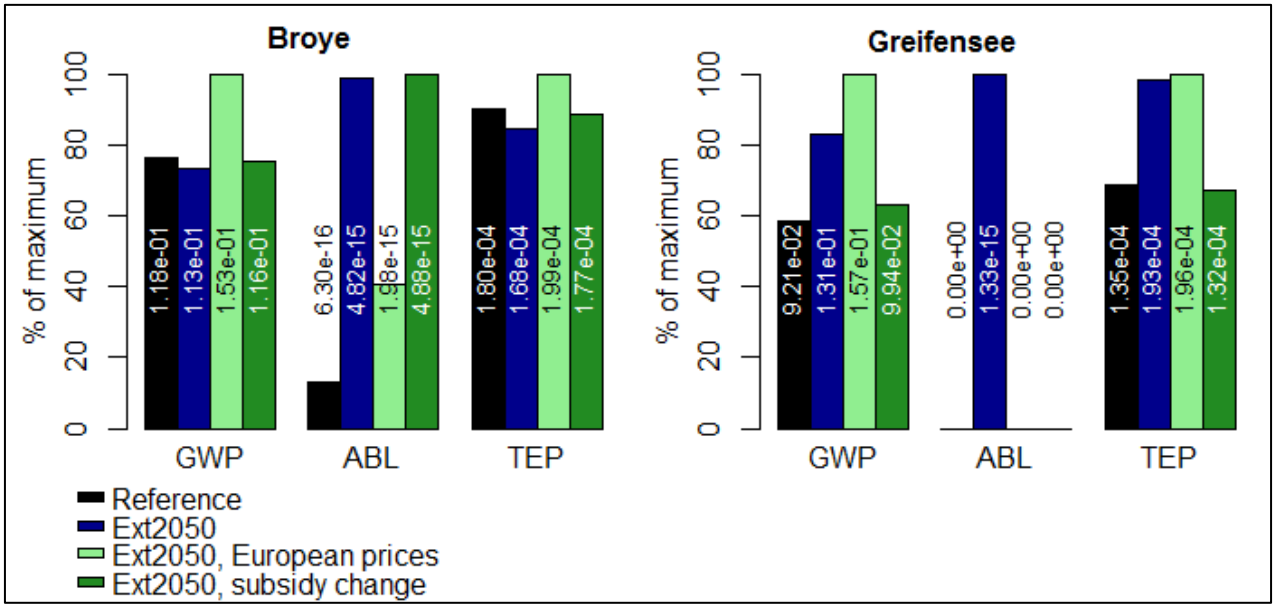


Figure 21: *mixed farm: environmental impacts of optimized farm for different policy scenarios (European prices and changes in subsidies, see Table 7 and the corresponding section in Chapter 4.1.3) under future climate (extreme change signal), per MJ dig. en (relative to the maximum). The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

For mixed farms, a decrease in prices causes a decrease in aquatic biodiversity loss of 60% in the Broye region and 100% in the Greifensee region, an increase in global warming potential of approximately 30% in both regions, and an increase in terrestrial ecotoxicity potential of 20% in the Broye region, 2% in

the Greifensee region. A decrease in subsidies has little influence in the Broye region compared to the corresponding future situation without subsidy changes "Ext2050", but causes a decrease in impacts in the Greifensee region.

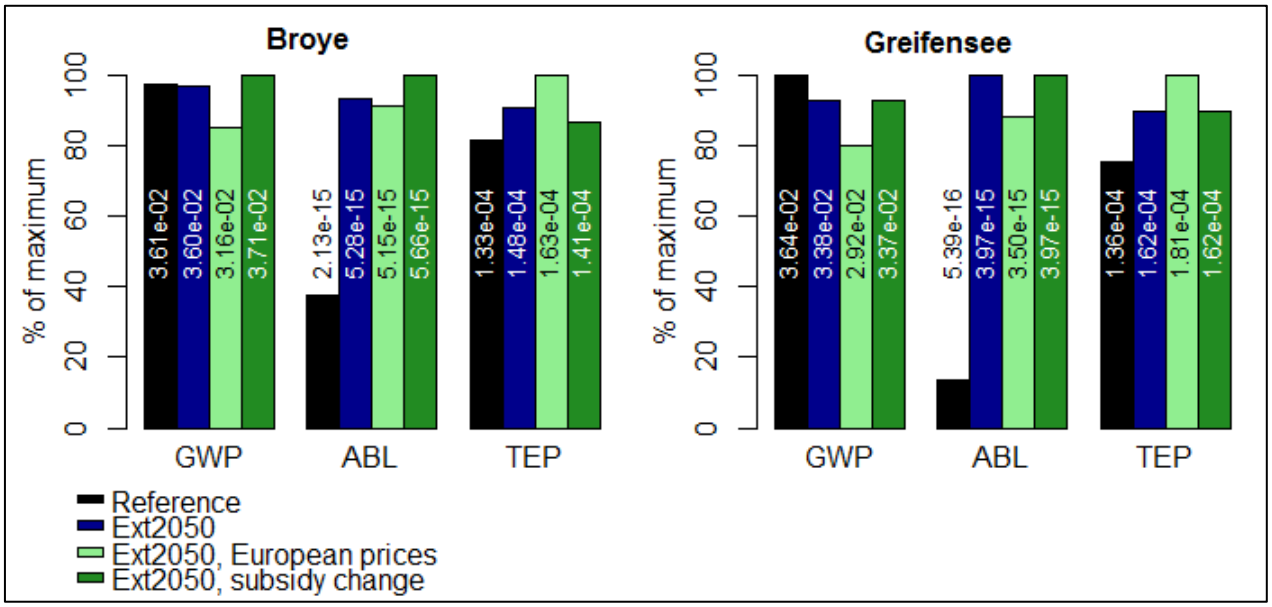


Figure 22: *crop farm: environmental impacts of optimized farm for different policy scenarios (European prices and changes in subsidies) under future climate (extreme change signal), per MJ dig. en (relative to the maximum). The values indicated in or above the bars are the impacts per MJ dig. energy: GWP = global warming potential (units: kg CO_2-eq./MJ dig. energy), ABL = potential aquatic biodiversity loss (units: GSE*y/MJ dig. energy), TEP = terrestrial ecotoxicity potential (units: kg 1,4DB-eq./MJ dig. energy).*

Crop farms are less sensitive to price and policy scenarios than mixed farms. Aquatic biodiversity loss remains above the reference level for all cases. A decrease in prices leads to slight decreases in global warming potential and potential aquatic biodiversity loss, and a slight increase in terrestrial ecotoxicity potential, for both regions (compared to the future situation without price changes). A decrease in subsidies on the contrary leads to a slight increase in global warming potential and potential aquatic biodiversity loss, and a slight decrease in terrestrial ecotoxicity potential in the Broye region, but to no significant change in the Greifensee region.

4.4 Interpretation

4.4.1 Impacts of climate change and economic optimization

For mixed farms which do not optimize profits under the future climate, the increase in relative impacts compared to the current situation (varying between 0% and 100%), is mainly due to a decrease in

production of approximately 10% (Broye) to 20% (Greifensee). This non-linear relationship can be explained by threshold behavior in several of the causal relationships (e.g. soil water content thresholds for irrigation application). Climate change in the Broye region enables an increase in production if the farm is optimized; this increase in production is proportionally higher than the corresponding increase in global warming potential (caused by an increase in fertilization intensity), and therefore the efficiency is improved. In this case, climate change mitigation and economic adaptation are compatible.

An increase in irrigation is the main adaptation undertaken to maintain yields in the future climate, in order to compensate the decrease in precipitation and increase in temperature during the growing period. This leads to the predicted increase in aquatic biodiversity loss (which is proportional to the amount of water consumed for irrigation, for marginal changes in withdrawals within a same region).
In the Greifensee region, economic optimization does not lead to mitigation of impacts. This is explained by the drop in production in the future climate: indeed the increase in temperature in this location, which is already warmer than the Broye region, is unfavourable for the yields of most of the crops modeled. In this case, climate change has a negative impact on the economic profitability of the farm, its productivity as well as its environmental impacts (see Table A3.3 in the Annex for functional units of profit and production).

For crop farms, without adaptation to climate change, a similar effect is observed as for mixed farms: productivity decreases by 10% (Broye) to 20% (Greifensee), whereas relative impacts increase by 5% to 75%. With adaptation to climate change, the increase in aquatic biodiversity loss and terrestrial ecotoxicity potential is due to a drop in production in both regions, caused by an extensification of the system (including a decrease in fertilization intensity), as well as an increase in irrigation in the case of aquatic biodiversity loss. This extensification thus causes a drop in production that is larger than proportional to the drop in these two impact categories. However, this decrease in production is compensated by a more-than-proportional decrease in the global warming potential: global warming potential per unit produced can thus be stabilized or even reduced through adaptation to climate change, showing that adaptation to climate change can simultaneously achieve mitigation of climate change impacts.

Thus for both farm types and regions, impacts generally increase in the future climate whether there is optimization or not, the main exception being global warming potential. In most cases, global warming potential can be mitigated by the economically optimized adaptation of the farm to the future climate,

unless the future temperature is too unfavourable for yields (as in the Greifensee region). Optimization in the future climate also maintains profits, or at least avoids significant decreases in profit (see the profit functional unit in Table A3.3). A major trade-off with the benefits of optimization is aquatic biodiversity loss, for both farm types and regions, which suggests a certain generality of this result. However, this outcome may also be dependent on the way the farm adaptation was modeled; a previous study using a different modeling approach (a water balance considering soil water content and the actual evapotranspiration to potential evapotranspiration ratio) also predicted increases of irrigation water requirements for the future climate in several regions in Switzerland [27], suggesting that water-use related impacts (assuming an extreme climate change signal) could well be relevant for other regions in Switzerland and possibly beyond in similar climatic regions.

4.4.2 Water restriction scenarios

The decrease in aquatic biodiversity loss is proportionally explained by a decrease in irrigation water use, in turn caused by the decrease in the profitability of irrigation (the variable costs increase in the case of water pricing, and the payback time of the capital costs increases in the case of water quotas). For both farm types, production decreases for both the water use restriction scenarios (by 15% to 21%), causing the observed increase in global warming potential and terrestrial ecotoxicity potential (relative to MJ dig. en.; the absolute values of these impacts decrease slightly, see Table A3.3).

Thus an increase in irrigation demand under climate change is only expected if water prices are negligible and availability is unconstrained [7]. The drawback of water use restriction is a decrease in productivity, which leads to the observed increase in the global warming potential and terrestrial ecotoxicity potential impacts relative to production. Thus, avoiding potential aquatic biodiversity loss is feasible through water pricing or quotas, but at the cost of other impact categories.

Although the relative reduction of potential aquatic biodiversity loss is up to 100%, whereas the relative increase in the other impact categories reaches only 20%, these numbers are not sufficient to determine whether this is a net gain or not. The absolute magnitude and acceptability of impacts in each category also need to be considered. Comparison of different impact categories is difficult, and requires normalization and weighting of the different protection areas. This can be ideally achieved through a broad stakeholder consultation and consensus-building process, as has been suggested for other examples of multi-criteria decision-making [143]. In the context of LCA, such normalization and weighting is often done by using endpoint impact categories (as explained in Goedkoop et al. (2009)

[26]); however, this would involve value choices and assumptions taken by the LCA modeler alone, and would not represent a broad consensus for the stakeholders concerned.

Finally, the implementation of water pricing or water quotas would require policy intervention: indeed, according to the farm optimization model, farmers show preference for using irrigation, as long as water is available and irrigation is profitable (although the decrease in profits for water pricing and water quotas is only 2%).

4.4.3 Sensitivity to climate scenario

The moderate climate change scenario is less favourable for yields than the extreme climate change scenario in the Broye region, where temperature increases approach the production optimum. As mentioned earlier, temperature exceeds the optimum in the Greifensee region for extreme climate change as well as moderate climate change. This difference can be explained by the background climate in each region: the Broye region is slightly cooler, therefore a larger increase in temperature (such as for "Ext2050") is more profitable to crop yields than a moderate increase in temperature (such as for "Mod2050").

The extreme climate change scenario involves decreases in precipitation of up to 30% (but only up to 24% in the Greifensee), whereas the moderate climate change scenario leads to decreases in precipitation of up to 10% only (and even increases in precipitation for all months except June for the Greifensee). This explains the difference in behavior of aquatic biodiversity loss between the two regions.

Thus climate change has a positive effect on yields in the Broye region provided it is strong enough (and irrigation is applied), whereas it has a negative effect on yields in the Greifensee region. The fact that impacts (except aquatic biodiversity loss) under moderate climate change in the Greifensee are higher than under extreme climate change for crop farms, but lower than under extreme climate change for mixed farms, was not explained in the available study which provided these adaptation scenarios [144], and is therefore hard to interpret here. Possibly, the increase in precipitation in the Greifensee region predicted by the moderate climate change scenario worsens conditions for crops, but less so for grasslands and pasture: mixed farms could therefore better cope with this climate change scenario.

For mixed farms, the adaptation strategy under the two climate change scenarios is similar for most aspects: according to Lehmann (2013) [144], the farm increases irrigation application and reduces fertilizer application under both the climate scenarios, and reduces land allocation to wheat production while increasing land allocation to sugar beet production. The strategy differs in that the decrease in fertilizer application is much higher under the moderate climate scenario in the Broye than under the extreme climate scenario, less irrigation is applied under the moderate climate scenario, and more land is allocated to sugar beet production (whereas under the extreme climate scenario, land is additionally allocated to rapeseed production).

In the Greifensee on the other hand, the decrease in fertilization application is much higher under the extreme climate scenario. Again this is explained by the fact that only moderate climate change in the Broye doesn't lead to as high a gain in potential yields (temperatures do not increase enough, but precipitation decreases slightly during the summer), whereas extreme climate change promotes higher potential yields (due to higher temperatures). In the Greifensee, which is already warmer than the Broye and beyond ideal temperatures, an increase in temperature is in any case deleterious to potential yields; and climate change induces an increase in temperature in both scenarios.

For crop farms, the trends in land allocation amongst crops is similar to that of mixed farms. Irrigation application likewise increases for both climate change scenarios, albeit to a lesser extent for moderate climate change. However, fertilization application increases under moderate climate change, whereas it decreases under extreme climate change. This might be explained by the above-mentioned disadvantages of moderate climate change for arable crop yields: farms which cannot rely on grassland and animal production may try to compensate the disadvantage of only moderate temperature changes by an increase in fertilization.

In summary, the choice of climate change scenario has a significant influence on the impacts; nevertheless, both climate change scenarios lead to an increase in impacts in the majority of cases compared to the reference situation.

4.4.4 Sensitivity to price and subsidy scenario

For mixed farms, with lower product prices, crop production becomes more extensive with much lower yields; irrigation in particular also decreases due to a decrease in profitability. More surface is allocated to grasslands and pastures, which are still intensively fertilized. This leads to an increase in global

warming potential per MJ dig. en. (due to N_2O emissions from organic fertilization of grasslands), despite the decrease in fertilization of the arable crops.

In the Greifensee region, the change in subsidies provokes an increase in intensity but also in yields of certain arable crops in the mixed-farm optimization model, and land allocation therefore shifts to these crops (i.e. wheat production is replaced by rapeseed); this allows overall production as MJ dig. en. to increase compared to the future situation without subsidy change. Impacts relative to overall production as MJ dig. en. are therefore similar to the reference scenario, and lower than the impacts relative to production for the situation under climate change only (where production decreases).
The assumed change in subsidies affects primarily mixed farms, since its principal component is a decrease in subsidies per animal capita (alongside some redistribution of crop subsidies among the different crops).

For crop farms, neither the modeled policy changes nor subsidy changes are sufficient to counteract the effect of climate change on water requirements and aquatic biodiversity loss: the gain in revenue thanks to irrigation remains worthwhile in the case of decreased subsidies, since revenue depends more on crop productivity and less on subsidies; crop productivity is in turn supported by irrigation. The gain in revenue allowed by irrigation is still worth it, even in the future climate with lower product prices.

In summary, impacts are more sensitive to the price scenario than the subsidy scenario; crop farms are less sensitive than mixed farms. A decrease in prices consistently causes a decrease in aquatic biodiversity loss in the future climate, however global warming potential may either increase or decrease according to the farm type.

4.4.5 Uncertainty

Differences observed between the calculated impacts of the scenarios modeled here must be set in relation to the uncertainty of the results. These have different sources: input data and parameter uncertainty, model uncertainty and uncertainty of assumptions and choices [145]. Uncertainties in some input data (e.g. climate scenarios) and assumptions (e.g. prices) have been addressed in the form of scenarios, which must not be seen as a confident prediction of future conditions, but rather as a range of possible situations, to which the sensitivity of the outcome can be evaluated. Conclusions which tend to be consistent over a variable range of scenarios may be considered to be more robust [146]. The models used (e.g. crop growth model used in the farm optimization model (see Annex A3.1 for description),

economic model, LCIA models etc.) are simplifications of real processes, relying on a limited set of parameters to simulate outcomes. The uncertainties of these models could be assessed using alternative models and comparing the outcomes. In general, marginal differences in impacts between scenarios should not be considered as representative.

Furthermore, possible sources of variability [145] were not entirely considered: limited spatial and temporal variability were considered (e.g. use of two regions and two periods of time, modeled as an average of a 25 year time series), as well as variability in the object type (e.g. type of farm). Variability in farmer behaviour was only considered in a limited way, since it was either assumed to be the economic optimum or the current practice.

4.4.6 Plausibility and variability

A prior study of whole-farm LCA impacts of real farms in Switzerland [22] provided impacts per ha*y for the categories global warming potential, terrestrial ecotoxicity potential, non-renewable energy use and eutrophication potential as N (nitrogen) equivalents, for all farm types. Some results were also available for dairy cow farm types in the lowlands (assimilated to the mixed-farm type addressed here). The range of impacts for these real farms were compared to the range of impacts of the farm scale scenarios assessed here (all scenarios), in order to verify the plausibility of models and input data. The results are compared in Table 10.

Table 10: *impact category ranges for real farms in Switzerland and results of the farm scale scenarios assessed here, for global warming potential, terrestrial ecotoxicity potential, non-renewable energy use and eutrophication potential as nitrogen equivalents. In black: results of the scenarios in the present study (all scenarios). In red: results of real Swiss farms (all farm types). In blue: where available, results of real Swiss farms (dairy farms in lowlands only). Average: average impact for all farms/scenarios considered; minimum: impact of farm or scenario with the lowest impact; maximum: impact of farm or scenario with the highest impact.*

Impact cagory	Average	Minimum	Maximum
Global warming potential (kg CO2-eq/ha)	5428 9000	4569 1500	6661 34000
Terrestrial ecotoxicity potential (kg 1,4 DB-eq/ha)	9 10 8	7 1	11 140
Non-renewable energy use	42858	38828	49295

(MJ-eq/ha)	50000 70000	10000	240000
Eutrophication potential (kg N-eq/ha)	55 130	41 20	63 450

In general, the order of magnitude of impacts between the two studies corresponds well. In particular, the average terrestrial ecotoxicity potential for all scenarios modeled here corresponds to the average of dairy farms in the lowlands shown in the reference study. Global warming potential, non-renewable energy resource use, and eutrophication potential related to N, obtained in the present study, were all situated around the lower quartiles of the reference study. Two possible explanations are proposed: (1) the farms modeled here correspond to optimized farms, where superfluous processing and inputs are avoided, whereas real farms are not necessarily optimally managed in this respect; additionally, above-average pest outbreaks, disease occurrences and extreme weather damage are not considered in the farms modeled here; (2) the modeled farms in the present study are simplifications of real farms, and do not include inventory flows which are uncommon in this farm type. Real farms present a higher diversity, and dairy farms may in reality not all be similar to the farm model used here. The economic farm optimization model does not account for certain variables which may have a significant influence on farmers' decisions and management options, for example pest control strategy, irrigation technology, further crops or animals, and also strategic farm orientation such as organic or extensive farming. Thus the variability in impacts observed between the scenarios assessed here is smaller than the variability observed between the impacts of real farms in Switzerland [22]. This suggests that although adaptation to climate change may make a significant difference on environmental impacts within a given region and a given farm type, other farm management strategies not considered here may lead to much larger differences in environmental impacts, exceeding the effects of adaptation to climate change.

4.4.7 Pesticide application

The relative terrestrial ecotoxicity potential is shown to increase in scenarios with more extensive production, due to the decrease in yields. The main contributor to terrestrial ecotoxicity potential in these scenarios is the application of pesticides on the farm. However the quantity of pesticides applied is not a variable related to yields in the model: the quantity and type of pesticides assumed to be applied vary only in relation to the surface allocated to each crop. In most scenarios, the absolute terrestrial ecotoxicity potential (not related to MJ dig. en.) actually decreases slightly due to the change in crop

mix. In addition, it can be expected that pesticide application will change in future due to changes in pest pressure, input and product prices, regulations, and according to the production intensity. According to the expert workshop conducted on the topic, for open markets and a more extensive production (e.g. the scenario "Ext2050, European prices"), it is expected that chemical pesticides will still be applied, but that their toxic effects will decrease somewhat thanks to a decrease in the quantity of pesticides used, and an improvement in their chemical composition. In case of open markets and a more intensive production, the amount and composition of chemicals applied is expected to remain similar to today; in case of high price pressure, toxic impacts could even increase due to use of cheaper and older chemicals (assuming there is no corresponding adaptation of pesticide regulation). For protected markets and a more extensive production (similar to the current situation, and possibly the scenarios "Ext2050", "Ext2050, subsidy change" and "Mod2050"), it is expected that in future, less chemicals will be applied, and will have less toxicity thanks to technological developments (indeed price protection enables use of less toxic and more expensive methods, while a decrease in productivity reduces the application intensity). It can be supposed that more profitable crops will be treated more intensively, whereas less profitable crops will be treated more extensively. In case of a more intensive production, alternatives to chemical pest control methods could become profitable, also leading to a decrease in toxic effects. Thus the modeled increase in terrestrial ecotoxicity potential relative to the production may be compensated in reality by an adaptation of the pesticide application strategy to the intensity of production. According to the expert workshop outcomes, a general decrease in the toxicity of the pest control methods can be expected in most of the scenarios.

4.4.8 Stakeholder feedback

The results of both the economic optimization and the environmental impacts for the Broye region were presented to a local stakeholder group, composed principally of farmers, and local and national administration representatives for agriculture and environment, during a half-day workshop. The objective was to obtain their opinion on the feasibility of the adaptation options and the desirability of their consequences. Three main points were drawn from this workshop: (1) large increases in irrigation are seen to be of limited feasibility, due to the important infrastructure they would require, the low probability that society would be willing to provide the necessary subsidies, and due to the limited amount of water resources available in the region. (2) Large changes in crop mixes, or shifts from arable crop production to livestock production were not considered realistic either, due to reluctance of farmers to change. However stakeholders generally considered this with a relatively short-term perspective, where large and rapid changes are not desirable. However, in the long term, if pressure

becomes sufficiently large, such changes may nevertheless become more acceptable. Maintaining the productivity of existing crops in a region relatively favourable for arable crops is seen as an important contribution to the self-sufficiency of Switzerland. Livestock farming also reduces farmers' living quality (due to less flexible working hours for example) and requires important capital investment. (3) Adaptation options to climate change foreseen by the stakeholders included step-wise technical solutions, such as the use of new cultivars with higher resistance to drought, and improvement of irrigation techniques to use water more efficiently. Large changes of the entire farm strategy were not prioritized. These technical solutions were not considered among the management variables of the economic model, due to high uncertainties in the prediction of cultivar improvements, and the focus on strategic changes rather than incremental changes [7]. Therefore it seems more likely that voluntary adaptation to climate change by the farmers may follow such step-wise trends in the near future; larger changes such as those assessed here may require policy intervention in order to motivate their application (if desired).

4.4.9 Extrapolation

The farms assessed here are representative for two locations in the Swiss lowlands, and two farm types. This gives an indication of the variability that can occur for different situations. Conclusions which are consistent for both farm types and both regions may be considered more robust, however not all conclusions are common to both locations and farm types: therefore extrapolation of these results to other regions and farm types is not recommended, because it is not scientifically supported. Nevertheless, as mentioned in Chapter 4.4.1, the importance of water use-related impacts could well be relevant for other regions in Switzerland, and possibly also other regions with similar climatic conditions.

The environmental impacts of the different scenarios according to three impact categories (global warming potential, aquatic biodiversity loss and terrestrial ecotoxicity potential) are shown. According to the correlation matrix (Table A3.4 in the Annex), the other impact categories initially assessed correlate strongly with global warming potential (namely non-renewable energy use, ozone formation, acidification potential, terrestrial eutrophication potential, marine eutrophication potential, freshwater eutrophication potential, human toxicity, aquatic ecotoxicity, land competition, and reduction of potential terrestrial biodiversity). This is not always observed in LCAs of agriculture [19, 20], and can be explained by the low variability of impacts between the scenarios assessed here. In the present study, the behaviour of these impact categories in the different scenarios can be extrapolated from that of

global warming potential (however the absolute gravity of an impact category cannot be extrapolated from that of global warming potential; absolute impacts for all initial indicators are given in Tables A3.5 and A3.6 in the Annex). The correlation of human toxicity and aquatic ecotoxicity with the global warming potential group rather than terrestrial ecotoxicity can be explained by the dominant contributors of these impact categories: for mixed farms, the dominant contributor to terrestrial ecotoxicity is onsite pesticide application (related to the crop mix), whereas for aquatic ecotoxicity and human toxicity, the dominant contributors are offsite non-pesticide flows related to cattle inputs for herd replenishment and feed inputs in particular. These are also the dominant contributors to global warming potential. For crop farms, terrestrial ecotoxicity is similarly dominated by onsite pesticide application, whereas aquatic ecotoxicity and human toxicity are dominated by offsite non-pesticide flows related to energy carriers, fertilizers and machinery. It must also be mentioned that the methods for estimating toxicity potential used here contain large uncertainties [147].

4.4.10 Limitations

The livestock module of the farm optimization model is considered by the developer as not very stable (small changes in assumptions lead to very different outcomes) [148]. Additionally, capital costs (such as infrastructure costs) were not included in the economic optimization. The farm scenarios used assume that farmers behave in an economically optimal way; this may however not be the case in reality, where other factors such as culture, tradition, peer pressure, and social and esthetical values, may also influence what decisions farmers actually take.

In this study, the environmental impacts are related to the productivity of the farm in terms of MJ dig. en. for humans. However other nutritional goals, such as provision of proteins and micronutrients, are not considered: this may disfavour animal products for example, since they typically have a lower calorie density. The use of a combined functional unit reflecting other nutritional goals could provide a more balanced evaluation, although existing studies show that this tends to further disfavour animal products due to their low content in other micronutrients [149, 150].

Furthermore, a decrease in productivity within a region may lead to an increase or decrease in relative and absolute impacts of the region itself, but it also implies a shift of production to exporting areas, in order to replace the lacking products. These areas may have different production impacts, and imports may also cause additional impacts due to transport for example. These consequential effects are discussed more extensively in a qualitative way in the general discussion section (Chapter 6.1.2).

The uncertainties mentioned in Chapter 4.4.5 also represent limitations of the reliability and applicability of the results, along with the limitations in extrapolation of the results to other situations mentioned in Chapter 4.4.9.

5 LIFE CYCLE ASSESSMENT OF REGIONAL ADAPTATION SCENARIOS

5.1 Goal and scope

5.1.1 Goals

This chapter presents the life cycle assessment of a selection of regional agricultural adaptation strategies developed in the AGWAM project [8, 9], with a focus on the mitigation of the cumulative regional impacts through regional measures. The environmental impacts of two selected water management scenarios are also addressed in this chapter: riparian shading of the river, and groundwater use for irrigation.

The agricultural adaptation strategies optimize regional agricultural land allocation and management in order to maximize the productivity while minimizing nitrate leaching, erosion and irrigation water use. For a description of the regional optimization model that served to produce these scenarios, see Annex A4.1 and Klein (2013) [151].

This chapter seeks to:

- Evaluate the environmental impacts of regional agricultural adaptation strategies to climate change for one case study region in Switzerland, allowing identification of impacts of concern when aggregated beyond the farm scale to an entire region.
- Identify an optimal environmental indicator set in this context and identify trade-offs between environmental indicators.
- Assess to which extent impacts of climate change and agricultural water use on aquatic biodiversity can be mitigated at the watershed scale by water management strategies (i.e. use of groundwater for irrigation, or riparian shading of the river system), and which are the most efficient strategies
- Provide an application example for the methods developed in Chapter 2 (results in Chapter 5.3.2) and Chapter 3 (results in Chapter 5.4), in the context of regional water management.

5.1.2 Case study region

The Broye region was retained for the detailed analysis conducted in this chapter, since it presents the most interesting case for assessing water-related management options: indeed, the climate change predictions for this region show a more severe increase in temperatures and decrease in precipitation

than for the Greifensee region; irrigation requirements at the farm scale were shown to be important (Chapter 4.3). The Broye region is described in Chapter 4.1.2.

5.1.3 Scenarios and temporal system boundaries

The scenarios assessed at the regional scale are summarized in Table 11 and are explained in the following paragraph:

Table 11: *summary of scenarios considered at the regional scale.*

Scenario name	Description
Reference	Reference situation under current climate, similar to real current situation
Ext2050 without adaptation	Same management as in the reference, but under the climate in 2050 (assuming an extreme climate change scenario)
Ext2050, productivity	Management maximizing the productivity (in terms of on-field yield of agricultural products, in kg) under the climate in 2050
Ext2050, preservation	Management minimizing erosion, nitrate leaching and water use under the climate in 2050 (thus maximizing the preservation of the environmental resources soil, groundwater and surface water)
Ext2050, compromise	Management for a weighted compromise between productivity and preservation
Ext2050, productivity, groundwater	Same scenario as in "Ext2050, productivity", but a part of the irrigation water is sourced from groundwater instead of river water.
Ext2050, productivity, shading 0.5	Same scenario as in "Ext2050, productivity", but the riparian shading factor of the river is increased to 0.5 (see Chapter 3.2.6).

The reference scenario in this case is not a fully optimized solution: it reflects the current management of land use, crop shares, fertilization intensity, irrigation intensity and tillage strategy (based on statistical data), and optimizes only the crop rotation spatial distribution [9]. This differs from the reference scenario used at the farm scale, which is an optimized solution (see Chapter 4.1.3): indeed, only an optimized scenario was available at the farm scale, whereas only a non-optimized scenario representative of current management was available at the regional scale. Optimization under the current climate did not provide a reference similar to current management at the regional scale [9], whereas this was true at the farm scale [7]. Reference scenarios at both scales can therefore be

considered as representative of the current situation, despite the fact that they are modeled in a different way.

A set of three strategies identified in the regional optimization model was considered for assessment here: maximizing productivity, minimizing impacts, and a compromise between both that maximizes the aggregated optimization score. These strategies are defined by certain weighting combinations of the multiple optimization objectives used at the regional scale (i.e. maximizing productivity, minimizing nitrate leaching, minimizing erosion and minimizing irrigation are each given a different relative weight [9]; this weighting represents the dominant goals to be achieved by the optimization). The strategies were identified using a cluster analysis, that accounted for the robustness of management within a cluster in order to achieve the desired outcome, and robustness of that outcome under different climate change scenarios [9].

Only the extreme climate change scenario is considered here (and is the same as described in the farm scale approach, Chapter 4.1.3). Indeed here sensitivity analysis was no longer the focus, but rather the assessment and mitigation of risks associated with a "worst-case" situation: aggregated impacts within the whole region, and extreme climate change. Thus in the face of an uncertain future situation, the maximal management requirements can be identified and the potential of options to address the worst case can be assessed. In addition, the management options identified by the regional optimization model are robust under both extreme and moderate climate change scenarios, and are therefore not expected to change between two such climate scenarios [9].

The groundwater scenario investigates a possible environmental impact mitigation strategy where a part of irrigation water is sourced from groundwater instead of river water. It was assumed that a part of the water requirements were sourced from the local aquifer during the months of July and August (during which irrigation demand is highest), to the extent that the latter could provide these requirements.

The riparian shading scenario investigates a further possible environmental impact mitigation strategy, where all irrigation water is withdrawn from the river, and riparian shading of the river is increased to a maximal feasible extent in order to mitigate changes in river water temperature.

5.1.4 Spatial and physical system boundaries

The LCA focused on the crop production level following the SALCA methodology for crop LCA [12, 19]: indeed, the scenarios of agricultural adaptation to climate change at the regional scale focused on land use allocation to crops; aspects specific to whole farms, such as infrastructure and farming units were not included as variables in the modeling of these scenarios; livestock is considered with a fixed composition and number, which do not vary with the scenarios, and is not explicitly allocated to a specific location in the region (and is therefore not spatially explicit) (see Annex A4.1 for description of the regional optimization model). The system boundary is cradle-to-gate (thus including inputs, but not processing and consumption of products after the farm gate). Internal transport was however not considered, since the land use allocation in the regional model did not respect realistic farm structural constraints, and therefore any assumption of transport from one farm to another was not possible (the consequences of this assumption are discussed in Chapter 5.6.4). Emissions from livestock are included, and are estimated using the relevant animal module for the SALCA farm methodology.

5.1.5 Functional units

Similarly to the farm optimization scenarios (see Chapter 4.1.5), megajoules digestible energy for humans (MJ dig. en.) were used as a functional unit of the regional optimization scenarios. The agricultural land occupation in ha*y did not vary among the scenarios, therefore the FU ha*y was used to represent the cumulated impacts in the region: indeed at this scale, impacts may accumulate, causing significant regional problems, which the relative impacts per MJ dig. en. would not illustrate adequately [10]. The monetary functional unit (CHF) was not assessed here since it was not part of the modeling approach. This combination of global and local impact categories with two functional units provides an overview of direct (local) and indirect (global) impacts, as recommended when planning land use policies [152]. Statistical modeling was done using the R software [83].

5.1.6 Environmental indicator selection

The initial set of environmental impact categories evaluated is the same as for the farm scale assessment (see Chapter 4.1.7 and Table 8), since this is the best assumption available for farming systems also at the regional scale. However, less scenarios were assessed at the regional scale (6 scenarios for the Broye region, see Table 11) than at the farm scale (in total 26 scenarios: seven climate and policy scenarios, most of which were assessed for two regions and two farm types, see Chapter 4.1.3). Therefore the reduction of the number of indicators to a subset using principle component analysis (see Chapter 4.1.3) was no longer possible (PCA requires a higher number of scenarios than of indicators in order to function). However the indicators selected using PCA at the farm scale are not

necessarily the best selection for the regional scale too, due to the differences in the variables and optimization modeling. Therefore a correlation analysis of impacts per MJ dig. en. (see Table A4.3) was used to identify a subset of indicators which minimize redundancy and keep trade-offs at the regional scale, taking stakeholder interest into consideration for the final selection.

5.1.7 Groundwater scenario: method and indicators

For this scenario, a groundwater model was established for the Broye region by Gomez (2012), in a master thesis co-supervised by the author of the present thesis [153]. This provided a water balance and groundwater head for the climate change and irrigation scenarios assessed here. The aquifer extent is shown in Figure 23.

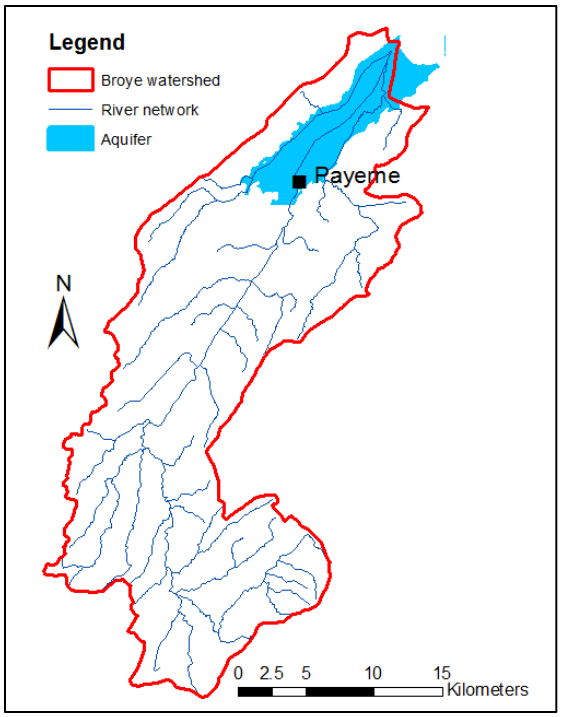


Figure 23: *alluvial aquifer extent in the Broye watershed (adapted from Gomez 2012 [153]).*

For the scenario assuming a productivity strategy in 2050 under extreme climate change ("Ext2050, productivity") and partial withdrawals from the groundwater, the water was assumed to be withdrawn from 10 pumping wells (located close to the irrigation perimeters, with suitable local aquifer properties and as far away from the river as possible in order to avoid effects on the base flow). The amount assumed to be withdrawn from groundwater was 18.5 mio m^3/y, from May to September, and mainly during July and August when irrigation demand is highest (this is only half of the total irrigation water required for agriculture in the scenario "Ext2050, productivity": indeed, the aquifer cannot provide the

total irrigation water demand, due to its hydraulic characteristics. The rest of the water requirement is assumed to still be withdrawn from the river).

The groundwater model served to predict the decrease in flow of water from the aquifer to the river (thus resulting in an impact on aquatic biodiversity), and the drop in groundwater table, which was then coupled with an existing LCIA method for groundwater withdrawals by Van Zelm et al. (2011) [36], which estimates the potential impacts on terrestrial biodiversity caused by the consumptive use of groundwater. This impact assessment method links the drop in groundwater head with an effect on terrestrial plant biodiversity in the form of potentially disappeared fraction of species. The available effect factor was developed for the Netherlands, but is assumed here to be valid for Switzerland too, since it was developed for plant species of temperate climate zones in Europe.

5.2 Life cycle inventory

5.2.1 Method

The LCA of agricultural regions is an approach for which no conclusive methodology exists yet [17], with few and only very recent examples of application [18]. Therefore several adaptations to the farm scale LCA had to be made for this thesis. The regional optimization model provided scenarios of spatially explicit crop management at a resolution of 25 ha [9] (this entails 1710 spatial units for the Broye region). In order to conserve the spatial detail without having to calculate 1710 inventories for each scenario, inventories were pre-calculated for all possible combinations of management (i.e. tillage strategy, fertilization intensity level, crop rotation) and environmental conditions that are spatially relevant (i.e. slope, erosion class and climate humidity class) using SALCA Crop [12], from which an aggregated inventory for the whole region could be calculated for each scenario: the inventory of each occurring situation was weighted by its area of occurrence in the scenario, and these weighted inventories were then summed. This approach is illustrated in Figure 24, and explained in detail in the following paragraphs.

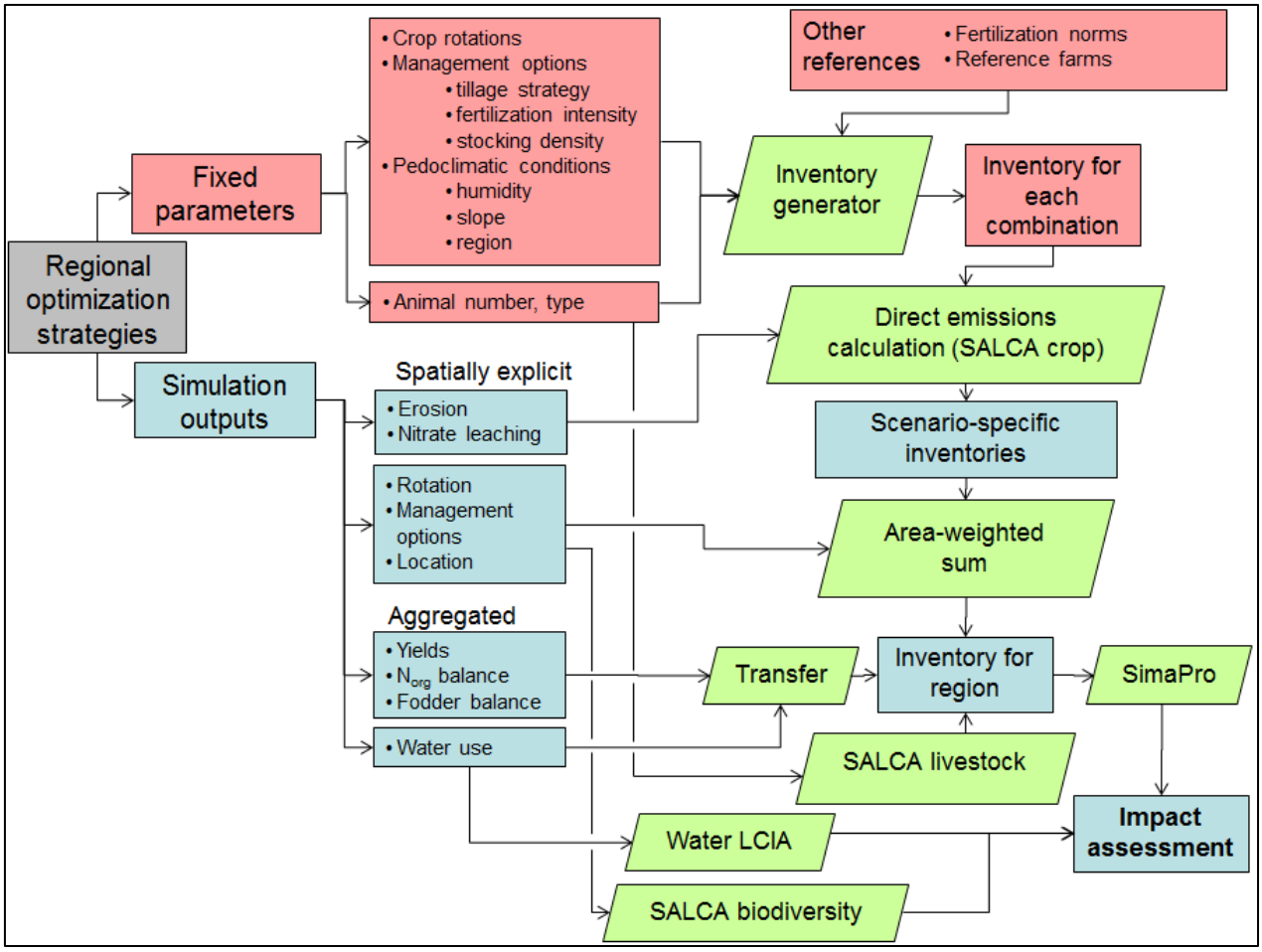


Figure 24: *work flow for the Life Cycle Assessment of the regional agricultural scenarios. The red constituents formed a fixed set of reference and categorical variables, serving to establish a set of reference inventories for each possible combination that can occur. The blue constituents are variables which change according to each strategy and scenario modeled: these serve to evaluate the emissions of each combination for the scenario-specific conditions. The green constituents represent calculation modules (implemented in Microsoft Excel and VBA, with the exception of SimaPro).*

The spatially explicit factors most relevant for the calculation of direct emissions were identified by a sensitivity analysis: the factors identified as spatially relevant were crop rotation, slope, erosion class (at risk or not), nitrate leaching (as provided by the regional optimization model [9]) and climate humidity class (classification of agricultural regions into dry-humid-wet zones, based on the climate suitability map of Switzerland [154]). The inventory of inputs and emissions was first established individually for each possible combination of spatially-relevant management variables (crop rotation, tillage strategy "conventional" or "no-till", and fertilization intensity level "low intensity", "medium intensity", "high intensity" [9]) and environmental conditions (erosion class "at risk" or "not at risk", and climate humidity class "dry", "humid", "wet") that could occur in the region. For inventory flows which were not available

from the regional optimization model (for example phosphorous, potassium and magnesium fertilizer application, pesticide application, machinery requirements), consistent assumptions were made based on inventories of farm reference models previously developed for Switzerland [22] as well as reference norms for Swiss agriculture [134]. Background inventory data was sourced from the SALCA database [140] and from the ecoinvent database [141]. This provided a database of pre-calculated inventories, from which the inventory of each combination occurring in a given scenario could be taken. The nitrate leaching and erosion of each pixel, which vary according to the scenario, were then integrated into these inventories of combinations occurring in the scenario. Direct emissions were then calculated using the SALCA models [137-139] (apart from erosion and nitrate leaching, already provided by the regional optimization model outputs). This produced a complete scenario-specific inventory for each combination occurring in the scenario. The area over which a given combination occurred in a scenario was determined from the outputs of the regional optimization model. This allowed aggregation of the inventories of occurring combinations for the whole region using an area-weighted sum: the inventory of 1 ha of a given combination was weighted by its total area of occurrence in a scenario, and added to the similarly weighted inventories of all other combinations. This produced an aggregated regional inventory. Inputs and emissions related to animals (such as fodder imports, methane emissions etc.) were added to the aggregated regional inventory (since they were not modeled in a spatially-explicit way). This resulted in the complete inventory for the whole region. The work flow for the regional scale LCA, as shown in Figure 24, was implemented in Microsoft Excel (2010) and the VBA programming language in order to enhance compatibility with existing SALCA tools (which are likewise implemented in Microsoft Excel and VBA). The inventories for all regional scenarios are provided in Annex A4.5, Table A4.4.

5.2.2 Assumptions for 2050

Assumptions had to be made concerning the irrigation technology, source of irrigation water, and the corresponding inventory flows. The reference irrigation strategy assumes sprinkler irrigation technology with an efficiency of 70% and water sourced from the river (as is currently the case in the majority of the region [112]). Pumping chambers and irrigation pipelines can be considered in the SALCA inventories. In future scenarios where the amount of water required exceeds the amount of water available in the river for a given month, water was assumed to be stored in reservoirs along the river (according to suggestions existing for the region [155]): thus the corresponding inventories include additional reservoir infrastructure (i.e. excavation of reservoirs and water-tight lining, using inventories available in ecoinvent).

5.3 Life cycle impact assessment and interpretation

5.3.1 Selected indicators

Four indicators were retained: three based on a correlation analysis of impacts per MJ dig. en. (Table A4.3) and one based on stakeholder interest. Global warming potential (GWP [23], for 100 years, expressed as kg CO_2 equivalents) again represents a category grouping most impacts, as in the case of the farm scale scenarios; potential aquatic biodiversity loss (ABL [128], expressed as GSE*y) again displayed an inverse correlation to most other categories; freshwater eutrophication potential (FWE [26], expressed as kg phosphorous (P) equivalents) also showed an inverse correlation to most other impact categories. Reduction of potential on-site terrestrial biodiversity represented a category of interest for the stakeholders (TBR, according to the SALCA-Biodiversity methodology [129], expressed as the reduction of the maximal biodiversity score achievable[1] or "points lost". This method was adapted for the present case in order to reflect only the relevant management variables). The calculation of impacts was implemented in the SimaPro software [133], except for reduction of potential terrestrial biodiversity and potential aquatic biodiversity loss, which were implemented in Excel VBA.

5.3.2 Environmental impacts of scenarios

Figure 25 and Figure 26 show the environmental impacts of the agricultural region, for the current reference situation as well as for four strategies under the climate in 2050: no change in management, maximization of production, maximization of natural resource preservation, and a compromise between the last two. The environmental impacts displayed are global warming potential, aquatic biodiversity loss, reduction in potential terrestrial biodiversity and freshwater eutrophication potential relative to the FU MJ dig. en., respectively FU ha*y. Absolute impacts for all investigated impact categories as well as both the FUs are available in Table A4.1. Results are interpreted in the following paragraphs, whereas a discussion of the plausibility of indicators not assessed in Chapter 4.4.6 is provided in Chapters 5.6.1 and 5.6.2.

[1] The maximum achievable score for this case study is 14.79 biodiversity points. The biodiversity score reflects the value of the habitat type and the management for various taxa, on a theoretical scale from 1 to 50.

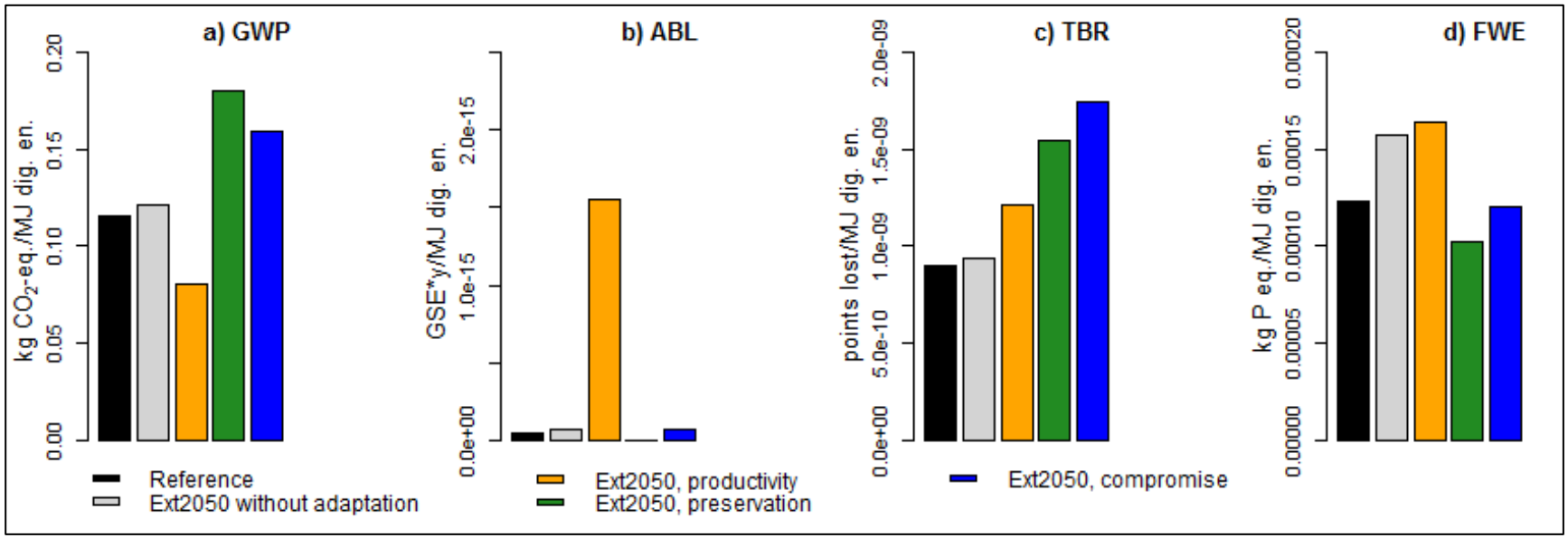


Figure 25: *environmental impacts per MJ dig. en. for the reference situation, and under climate change for 4 strategies: no change in management, maximization of production, preservation of natural resources, compromise; a) global warming potential (GWP), b) potential aquatic biodiversity loss (ABL), c) reduction in potential terrestrial biodiversity (TBR), d) freshwater eutrophication potential (FWE).*

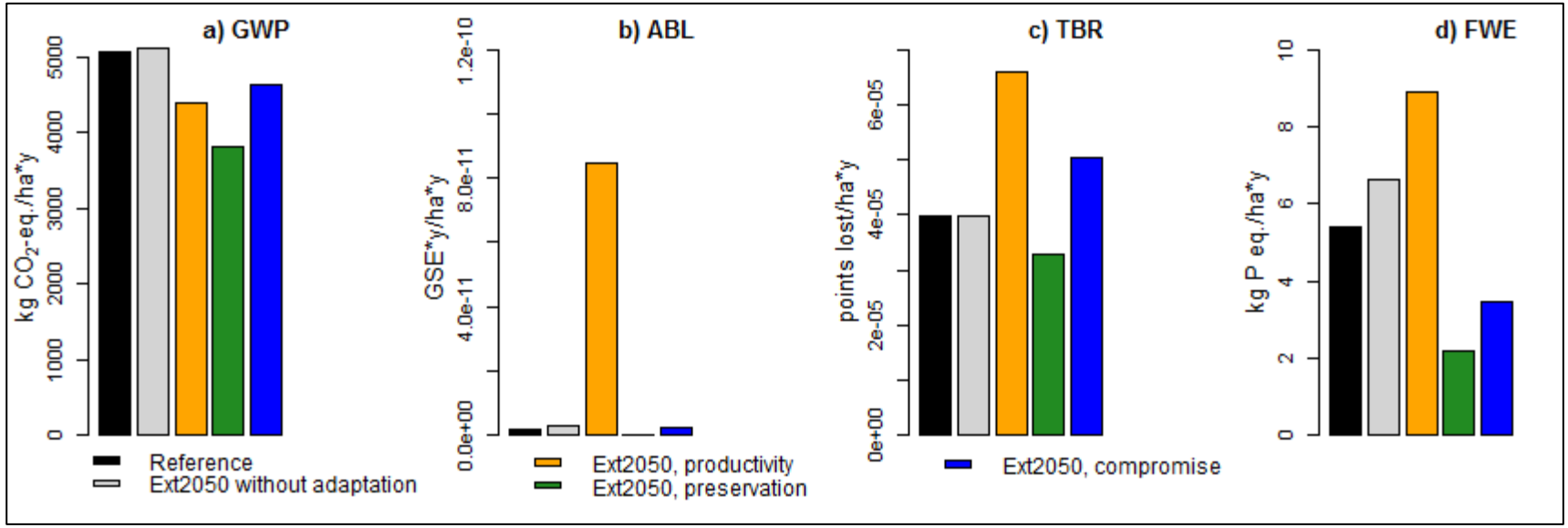


Figure 26: *environmental impacts per ha*y for the reference situation, and under climate change for 4 strategies: no change in management, maximization of production, preservation of natural resources, compromise; a) global warming potential (GWP), b) potential aquatic biodiversity loss (ABL), c) reduction in potential terrestrial biodiversity (TBR), d) freshwater eutrophication potential (FWE).*

In general, no adaptation under the future climate leads to very little change in the evaluated impacts, compared to the reference situation, except for freshwater eutrophication potential. Indeed management stays very similar (only the amount of irrigation water applied changes slightly), and yields decrease only slightly (see Figure A4.1). The larger change in freshwater eutrophication potential is due to a slight increase in nitrate leaching and erosion for this scenario, as predicted by the regional optimization model [8, 9]. The reason of these changes is not provided by the modeler [9], however it may be supposed that similarly to the farm scale scenarios, the increase in temperature is detrimental to yields: plants thus take up less nitrogen, of which more is therefore available for leaching. Erosion may be possibly affected by the slight increase in irrigation water application. In comparison, the outcomes of the farm-scale assessment in Chapter 4 suggested that impacts will clearly increase: it must be recalled that the regional scale assessment covers a multitude of different conditions beyond those of the single farm of the Broye region assessed in Chapter 4. Therefore some sub-regions and the farms therein may experience less severe increases in impacts than those assessed in Chapter 4. In particular, the yield at the farm scale (54'484 MJ dig. en./ha*y for the reference scenario) decreases by 9% under future climate without adaptation, whereas the yield at the regional scale (43'832 MJ dig. en./ha*y for the reference scenario) decreases only by 4% under future climate without adaptation. The initial yield at the regional scale is lower than the initial yield at the farm scale (indeed, under the current climate, the region around Payerne is warmest and most suitable for crop production, which leads to higher yields as MJ dig. en./ha*y). However the region around Payerne, which is already dryer and hotter than the hill regions, may be more severely affected by climate change than the hill regions; these in turn may even benefit somewhat from climate change: indeed, they become more suitable for crop cultivation, leading to higher yields as MJ dig. en./ha*y in this area. In summary, without optimization under the future worst-case climate (in other words, assuming "business as usual"), impacts are only slightly higher. The potential of adaptation scenarios can therefore be seen rather to improve the "business as usual" situation rather than avoid drastic increases in impacts caused by climate change.

Optimization strategies in future lead to very different outcomes concerning global warming potential: maximizing productivity leads to a decrease in global warming potential per MJ dig. en. due to a large gain in production, as well as a decrease in global warming potential/ha*y. This scenario entails more arable crops and less grasslands, the latter being the dominant contributor to global warming potential in this assessment, due to organic fertilizer application and ensuing N_2O emissions. Preservation of natural resources leads to a high global warming potential per MJ dig. en.: indeed, although this

scenario is more extensive and leads to lower global warming potential per ha*y (it has less input-intensive arable crops and more grasslands which are extensively managed, which greatly reduces the N_2O emissions), the large drop in productivity (measured as MJ dig. en. for humans) due to the decrease in arable crop production leads to higher impacts related to production. The compromise scenario sees an increase in global warming potential/MJ dig. en., and only a limited reduction in global warming potential/ha*y. Indeed, this scenario increases the share of grasslands at the expense of arable crops, but these grasslands are still fertilized at intensive levels, leading to sustained N_2O emissions. Hence this compromise scenario does not achieve mitigation of global warming potential, in contradiction to its stated goal (mitigation of environmental impacts while avoiding large drops in production [8]): this illustrates the importance of considering multiple impact categories when seeking to address the mitigation of environmental impacts. In summary, global warming potential/ha*y decreases in all optimized strategies in future, but yields also decrease except for the strategy maximizing productivity: thus only the strategy maximizing productivity enables a reduction of global warming potential related to production.

Potential aquatic biodiversity loss is a major trade-off for the gain in productivity and global warming potential efficiency achieved by the productivity strategy, due to the large amounts of irrigation water it requires under the future climate. Indeed, aquatic biodiversity loss increases both per MJ dig. en. and per ha*y: thus the increase in productivity comes at the cost of aquatic biodiversity, both in absolute and relative terms. This impact entails a potential loss of approximately 8% of fish and aquatic macro-invertebrate species within the Broye watershed itself compared to the future expected species richness (this loss corresponds to 1 fish species and 8 macro-invertebrate species), and a total of 1.81E-5 global species extinction equivalents (GSE*y). Climate change itself is expected to cause a decrease in the species richness of the watershed of approximately 13% compared to the current expected species richness, due to a decrease in average discharge (this corresponds to 2 fish species and 14 macro-invertebrate species). The other three strategies do not lead to an important increase in this potential loss (in particular, the compromise scenario, which aims to optimize productivity against nitrate leaching, erosion and irrigation water use, indeed achieves a consistently moderate impact on aquatic biodiversity).

The reduction in potential terrestrial biodiversity per ha*y is highest for the productivity strategy, due to an increase in fertilization intensity and in arable crop land use. However thanks to the gain in production, this remains the best strategy concerning reduction of potential terrestrial biodiversity/MJ

dig. en. The preservation strategy leads to a decrease in reduction of potential terrestrial biodiversity/ha*y (and is thus more favourable for terrestrial biodiversity) due to less intensive fertilization, more grasslands and more conservation tillage. However the loss in production causes an increase in reduction of potential terrestrial biodiversity/MJ dig. en. The compromise strategy has a relatively high reduction in potential terrestrial biodiversity/ha*y, due to its large share of intensively used grasslands (which are particularly unfavourable for the biodiversity score). In combination with its moderate productivity, this leads to the highest reduction in potential terrestrial biodiversity/MJ dig. en. Again this underlines that what may be an optimal compromise between productivity and nitrate leaching, erosion and irrigation water requirements is not necessarily favourable for other indicators of environmental impacts. In the present case, intensively managed grasslands enable a certain productivity without high demands of irrigation water, and are generally preferable compared to arable crops as concerns erosion and nitrate leaching. They are however very detrimental to on-site terrestrial biodiversity [129].

Freshwater eutrophication is highest for the productivity strategy (both per ha*y and per MJ dig. en.). It is mainly caused by direct field emissions due to erosion, runoff and leaching, rather than emissions in the background processes. This can be explained by the high fertilization intensity, high irrigation rate and high erosion occurring in this scenario, leading to high phosphorous availability and transport to water bodies. Both the preservation and compromise strategies lead to lower freshwater eutrophication than in the current situation, as well as the future situation without adaptation.

It should be observed that no single strategy simultaneously mitigates all impact categories; in particular, there is a trade-off between global warming potential and aquatic biodiversity loss due to the high irrigation levels required in order to improve productivity. This suggests that if productivity is to be maintained or even increased under the future climate while mitigating contribution to climate change, measures must be developed to mitigate or compensate the expected impact to aquatic biodiversity. This could include a change in irrigation technology (indeed, the scenarios assessed here assumed the use of sprinkler irrigation [9], which is not the most efficient). However, drip irrigation is typically used for vegetable cultivation, and may not be feasible for arable crops which are tilled for example. It may also involve high infrastructure costs. Other potential measures include sourcing the irrigation water from the lake rather than the river (however, this also entails high infrastructure costs, see Chapter 4.4.8); mitigation of impacts of irrigation water use by riparian shading of the river providing the water;

and sourcing of a part of the irrigation water from groundwater rather than from the river: the results of the latter two options are addressed in the following two chapters (Chapters 5.4 and 5.5).

5.4 Mitigation of aquatic biodiversity impacts through riparian shading

5.4.1 Water temperature change

Figure 27 shows the daily average water temperature throughout the year (modeled according to the method described in Chapter 3), averaged over the 20 year simulation period (these 20 years are simply repetitions of a year representative for 2050, considering natural variability expected in the climate in 2050, since the exact daily weather of 2050 itself cannot be predicted). The scenarios represented are "Reference", "Ext2050", "Ext2050, productivity", and "Ext2050, productivity, shading 0.5", and four temperature observation stations, representative of the different conditions occurring in the Broye watershed, are shown.

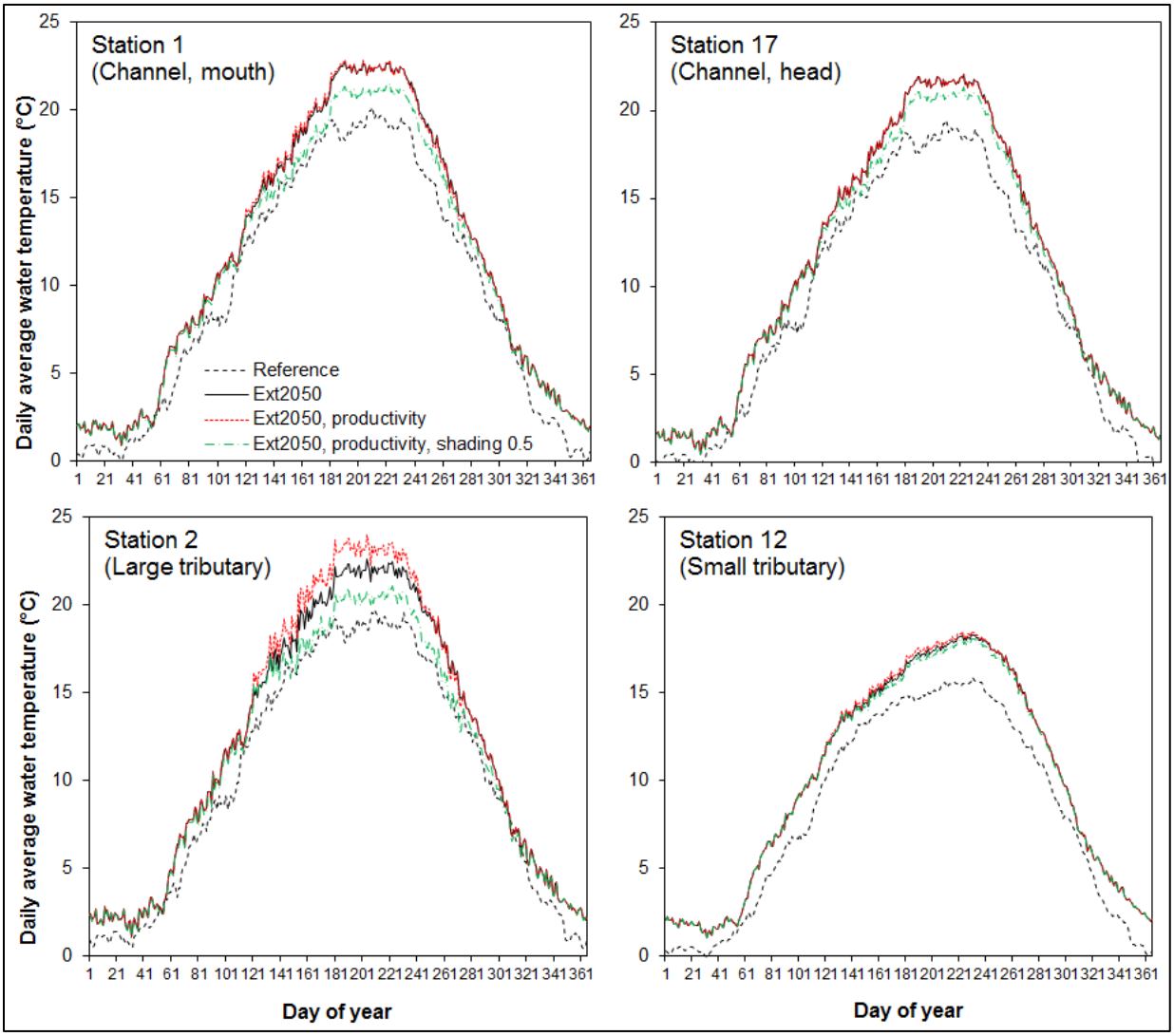


Figure 27: *daily average water temperature (average of the 20 simulation years representative for the year 2050) for the four scenarios considered, for a representative selection of stations.*

The variability of the daily average temperature is dampened here by the 20 year averaging (an example of variability within a single year can be observed in Figure 11 in Chapter 3.3.1). Stations 1 and 17 (both located on the main river channel, with station 1 at the mouth and station 17 at the head of the channel, see Figure 10) exhibit a similar behavior; water temperature tends to decrease slightly along an upstream gradient, corresponding with decreased upstream residence times, higher altitudes etc. The tributary stations (stations 2 and 12) exhibit lower peak temperatures than the channel station in summer: indeed, the first has a more favourable geometry (larger depth to width ratio) and shorter upstream residence time; the second has in addition more shading. The changes in temperature in the scenarios are larger than the uncertainty of the simulations (measurable with the root mean square

error provided in Chapter 3.3.1). The changes can thus be considered significant (since they show a consistent trend in one direction, whereas the model errors are assumed to be random and independent), and the observed trend in the expected value can be considered reliable, even though the actual expected value is uncertain.

Climate change causes an increase in water temperature throughout the year in all stations, due to the increase in air temperature. This increase is particularly marked in summer (on average 4.5 °C in the channel stations from June to August), when precipitation also decreases. The effect of averaged irrigation withdrawals between May and September (scenario "Ext2050, productivity") is marginal in most stations. Indeed, water levels are already below a certain threshold, beyond which the principal driving factor in this system is air temperature and solar radiation; further decreases in water levels no longer have a significant influence on water temperature. In the case of station 12, the amount withdrawn is also quite limited. Only station 2 (and station 4 in the same tributary, not displayed here) show a significant response to withdrawals in the future climate. This may be explained by a higher withdrawal to discharge ratio as well as a deeper and narrower channel: a reduction of depth still has an influence on water temperature in this case. This suggests that in the present case study, the effect of local anthropogenic interventions (such as river water withdrawal for irrigation) are marginal compared to that of global anthropogenic disturbances (such as climate change caused by greenhouse gas emission), when considering changes in river water temperature.

Shading of the entire network with typical riparian forest vegetation (equivalent to a shading factor of roughly 0.5 for the occurring channel widths) can significantly mitigate the increase in summer temperature due to climate change, with an average decrease in water temperature of 2°C during July and August at station 1, and 2.6°C during the same period at station 2. Stations 17 and 12 exhibit less sensitivity to shading: indeed, both are initially more shaded than stations 1 and 2: the gain due to an increase in shading is therefore limited. The riparian vegetation is assumed to be deciduous, thus shading only occurs in summer. Evergreen vegetation would be expected to cause in addition an increase in winter temperatures [156]. However, even this maximal feasible shading is insufficient to compensate the entire temperature increase caused by climate change, for all stations. For smaller streams however, shading may achieve a higher factor than 0.5 (such as 0.7), since the width of the channel is smaller and tree canopy can more fully cover it: thus shading may achieve a better result for these streams than as modeled here. For the channel and larger tributaries, further mitigation can only be provided through other complementary approaches, such as regulation of groundwater withdrawals

in order to enhance exfiltration of cooler groundwater into the river during the summer (see Chapter 5.5), or revitalization of the channel in order to provide more favourable sediment and bank conditions, as well as refuge areas for aquatic fauna during peak heat stress periods. Withdrawals from the tributary which is most affected (i.e. station 2) could be shifted to another water source (such as the nearby lake Morat).

In general, the efficiency of cooling measures depends on the driving factors of the system [157]; here, radiation and air temperature appear to play a significant role, and it is therefore not surprising that shading has a significant effect on water temperatures. Riparian shading may also provide the additional benefit of reducing potential eutrophication problems such as algal blooms [158, 159]. In any case, water temperature management should involve a holistic assessment considering relevant impact mechanisms on water temperature, such as discharge, shading, groundwater flow and channelization [160, 161]. The first three elements have been partly addressed in this thesis; an assessment of the effects of channelization and river restoration would provide a complementary perspective: indeed, it has been observed that river morphological variability generally supports aquatic species richness [162]. Ultimately, planning of efficient riparian restoration should integrate further aspects in addition to environmental criteria related to temperature, such as economic feasibility criteria [163].

5.4.2 Impacts of river water temperature changes

This chapter provides the results of the impact assessment relating the changes in water temperature (see the previous Chapter 5.4.1) to a potential impact on the aquatic ecosystem and aquatic biodiversity (using two impact assessment methods as explained in Chapter 3.1). This enables the identification of the potential impacts linked with water temperature changes, allowing an assessment of the efficiency of riparian shading as a mitigation measure for the impacts of river water withdrawals for irrigation on aquatic biodiversity.

The first impact assessment method used [70] compares a time series of daily average water temperatures with the temporally explicit temperature tolerances of an indicator species (considered representative of the typical faunal assemblage according to the type of river assessed), and estimates the deviation of the actual temperature regime from the expected natural temperature regime: a large deviation is assumed to lead to impacts on the expected faunal assemblage in the river. The temperature changes throughout the 20 year simulation period were found to cause no excessive change in the expected thermal regime for the identified river types of the Broye (epipotamal and

hyporhithral zones [164], which correspond approximately to the barbel and grayling fish zones respectively). With no change in the attributed grade "slightly modified" (on a 5-level scale of nearness to natural conditions: "natural", "slightly modified", "significantly modified", "strongly modified" and "unnatural/artificial"), the corresponding expected aquatic fauna would not be affected. However the Broye channel also exhibits some species belonging to metarhithral (kollin) river types (corresponding approximately to the trout fish zone); therefore the effects of the temperature changes were also assessed for this river type, revealing a slight degradation of the conditions for the corresponding aquatic fauna from "slightly modified" to "significantly modified", for certain years only. This indicates that for most species present, the expected temperature changes will be of little consequence, except for species with cool water preferences (such as those belonging to the metarhithral river type), which could be occasionally affected by the increase in temperatures. These may migrate further upstream towards cooler reaches, however the reaches upstream exhibit different characteristics (width, slope, depth) and may therefore not be an adequate alternative for all of these species. Thus although the species richness in the entire watershed may not change directly due to temperature changes, the species composition could shift along the temperature gradient [165]; overall species richness may be at risk if the entire habitat with adequate temperatures for certain species disappears [166]. It must be noted that this assessment only considers the average situation, and does not account for extreme situations which may occur (such as high withdrawals coinciding with extreme air temperature during a few days). The impact assessment approach used here also provides an assessment using daily minimum and maximum temperatures compared with life-stage specific temperature requirements of the indicator species. This would give an insight into the effects of extreme situations, however this approach was not used here due to excessive uncertainties (indeed, maximum daily irrigation is not considered as such in the temperature model used here; future predictions of daily extreme climatic data are unreliable and were not available for this study; and estimations of maximum daily water temperature from daily climatic series is very unreliable since it depends on the conjunction of several climatic factors which are themselves uncertain. Only minimum water temperature can be estimated relatively reliably, however this is probably not a critical issue in summer in this region).

The second approach for estimating the impacts of the change in water temperature on aquatic biodiversity used an existing effect factor (from Verones et al. (2010) [56]): this relates a change in water temperature to a potential loss of aquatic biodiversity. The effect factor is multiplied by the change in temperature found in this study (see Chapter 5.4.1) and provides a potentially disappeared fraction of species in the reach affected. According to this approach, the increase in annual average temperature

due solely to climate change was estimated to cause a loss of 0.20% of aquatic species (calculated using station 1, which is exposed to the accumulated withdrawals and decrease in discharge of the whole watershed). Water withdrawals in the "Ext2050, productivity" scenario lead to an additional loss of 0.03% of aquatic species. In comparison, the estimation of aquatic biodiversity loss due to decreased discharge (according to the method described in Chapter 2, see results in Chapter 2.3) forecasts a loss of 8.21% of local species in the watershed.

Together, these results suggest that the principal impact pathway of climate change and water withdrawal on aquatic biodiversity in this case study is probably not the change in water temperature directly, but rather the change in discharge in general, or a related indirect factor (such as the decrease in habitat volume, the decrease in contaminant dilution due to a decreased volume of the receiving water body, the decrease in connectivity and thus access to refuges due to lower discharges, or a change in the toxicity of contaminants due to a change in ambient temperature). Such effects can only be estimated using a more detailed reach-scale water quality modeling approach. This dominance of changes in discharge above changes in temperature in causing impacts on aquatic biodiversity corroborate previous results in a regional-scale study of the effects of discharge reductions in Michigan, USA [54]: changes in discharge were also found to have more effect on species than changes in temperature. One explanation proposed by the authors of the afore-mentioned study is that changes in temperature may only become relevant when exceeding species tolerance threshold values (which was manifestly not the case here for average changes in water temperature).

The impacts of extreme temperatures (rather than the average situation) are not captured by either impact assessment approach used here. However, high fish mortality has been observed in periods of peak temperatures and low flows in the Broye channel (based on own observations during temperature measurement campaigns), which may indicate that extreme situations play an important role in the impacts on aquatic biodiversity.

In summary, the contribution of climate change to changes in river water temperature does not ultimately lead to an important impact on aquatic biodiversity. The local withdrawal of water itself has a potentially higher impact on local aquatic biodiversity, but not through a change in water temperature. Thus management options should focus firstly on addressing the amount of discharge available in the river. Shading could represent an efficient complementary measure to offset peak stress situations.

5.5 Mitigation of aquatic biodiversity impacts through use of groundwater

5.5.1 Effects of groundwater withdrawals

This chapter assesses the effect of sourcing a part of the irrigation water demand for the scenario "Ext2050, productivity" from groundwater rather than from the river, with the view of assessing whether impacts on aquatic biodiversity can be mitigated (as described in Chapter 5.1.7). The model and scenario assumptions are described in Chapter 5.1.7.

The groundwater withdrawals incurred (with consideration of any return flow to the groundwater from the field) lead to a significant drop in the groundwater head compared to the situation without withdrawals in the future climate, during the most critical month (August), as well as on average during the year, as shown in Figure 28a and Figure 28b.

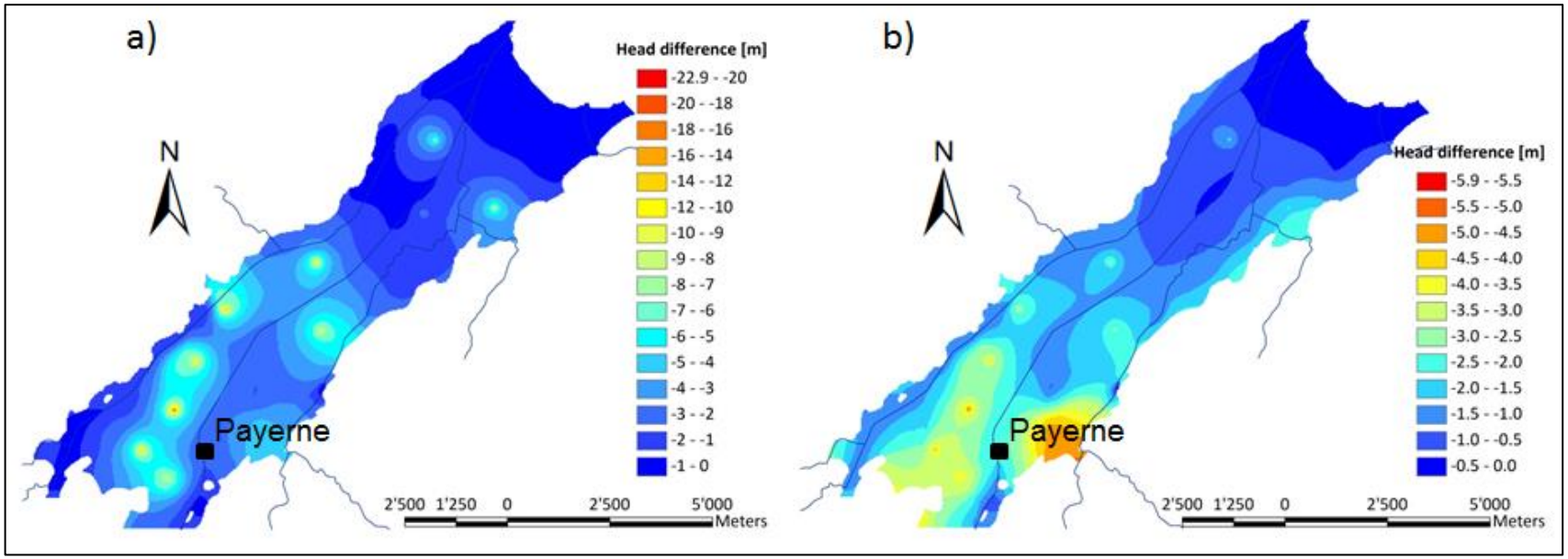


Figure 28: *groundwater head drop for the simulated withdrawals, compared to the future reference situation (groundwater level under future climate, without agricultural withdrawals), a) during the month of August, b) on average during the year (adapted from Gomez 2012) [152].*

The agricultural pumping wells are clearly visible at the locations of the highest head drop[2]. The agricultural groundwater withdrawals lead to a decrease in the water flow from the aquifer to the river: this flow drops from 0.73 to 0.312 m^3/s during the month of August (for which the base flow in the river is around 1 m^3/s in 2050), and from 0.80 to 0.71 m^3/s on average throughout the year.

This causes a decrease in river discharge: the impacts of this decrease can be assessed similarly to a river water withdrawal, using the LCIA method developed in Chapter 2. According to this approach, the decrease in discharge due to the reduction in flow from the aquifer (in turn caused by the agricultural groundwater withdrawals) leads to a potential aquatic biodiversity loss of 1.55E-6 GSE*y. This represents 8.6% of the impacts caused if all irrigation water is withdrawn from the river (1.80E-5 GSE*y, see Chapter 2.3.2). The total impact caused by the groundwater withdrawals for irrigation as modeled here, as well as the river water withdrawals for the remaining irrigation water demand, amounts to 1.22E-5 GSE*y, which is 67.7% of the impacts if all irrigation water is withdrawn from the river. Thus the use of groundwater for irrigation can avoid 32.3% or one third of the potential impacts on aquatic biodiversity.

However, the head drop caused by the agricultural withdrawals may cause burden shifting from aquatic biodiversity to terrestrial biodiversity. Indeed, the agricultural withdrawals of groundwater in the scenario assessed here lead to a potential loss in terrestrial biodiversity of 227'098 PDF*m^2*y, using the characterization factor of Van Zelm et al. (2010) and assuming an egalitarian perspective [36]. These estimated potential impacts on terrestrial biodiversity due to groundwater consumption probably overestimate the actual impacts to be expected, since a large majority of the land use above the aquifer is agricultural: thus there are hardly any natural terrestrial ecosystems that would actually be affected by the groundwater head drop.

Therefore the alternative sourcing of irrigation water from the local aquifer may avoid a part of impacts on aquatic biodiversity without dramatically increasing impacts on terrestrial biodiversity for this particular case. But it is insufficient to satisfy the full irrigation water requirements of the maximum productivity scenario in the future climate.

[2]The main domestic pumping well for the region was also modeled, and is visible to the east of Payerne; an increase in withdrawals from this well due to a linear increase in population extrapolated from past population growth, with a constant per capita water consumption, was assumed for the future situation. This influences the reference level of the groundwater, but does not influence the head drop measured due to the agricultural withdrawals only.

5.6 Discussion

5.6.1 Plausibility of biodiversity impacts

Concerning the potential loss of aquatic biodiversity: a previous study using the species-discharge relationship approach for two biogeographic regions in the USA predicted that decreases of 20% and 50% of river discharge would lead to a loss of 2 to 5%, respectively 6 to 13% of fish species in the rivers studied [78]. The total decrease in discharge due to climate change and irrigation in the productivity strategy amounts to 30% of the current discharge: the corresponding loss of fish species in the watershed is approximately 5%. This is consistent with the order of magnitude of the predicted loss in the afore-mentioned study.

Concerning the reduction of potential terrestrial biodiversity: the absolute biodiversity scores of the scenarios range from 0.7 to 7.7 (not shown in Figure 25-Figure 26; the initial reference score is 14.8, and the reductions in the score for each scenario are given in Table A4.1), whereas this score can theoretically range from 0 to 50. However, the maximum typically achieved in Swiss conditions (with the most favourable management and crop) reaches 30 for grasslands and 10 for arable crops [167]. The results of the scenarios modeled here are therefore plausible, but rather low.

5.6.2 Severity of total phosphorous emissions

The severity of the potential direct emission of phosphorous (P) in the catchment was assessed by comparing it with a rough estimate of phosphorous emissions from agriculture to surface waters in Switzerland. These were estimated to amount to 1.09 mio kg P per year for the whole country [31]. The direct emission of phosphorous for the productivity strategy amounts to 0.33 mio kg P per year: this would represent 30% of the total emissions from agriculture in Switzerland, whereas the agricultural area in the Broye watershed represents only 3.3% of the total agricultural area in Switzerland [109]. Even the direct emissions for the preservation strategy reach 14% of the total emissions. Agriculture in the Broye region is expected to be more intensive than the Swiss average (which includes typically extensive systems such as mountain grasslands) due its particular suitability for arable crops, however this may not be sufficient to explain the factor 9 between area ratio and phosphorous emission ratio. Phosphorous emissions as modeled in this study rely on several assumptions: (1) phosphorous is applied to fields assuming average phosphorous fertilizer application for each crop, corrected for yields [134]: this may be an overestimation of what is actually applied; (2) phosphorous emissions from the fields are modeled using SALCA [168], with the default assumption that 20% of eroded material is transported to

water bodies (Swiss average). This amount can vary from 5% to 25% in the SALCA method. However this was not found to be a highly sensitive spatial parameter in a prior sensitivity analysis, and it is therefore not expected that this explains the divergence from the Swiss average emissions. A further potential explanation may be that the Broye region contains a hilly area, which may be more exposed to erosion and thus exhibit higher phosphorous emissions to water than average agricultural land present in the flat lowlands.

The high phosphorous emission result may suggest that the adaptation strategies in the region could cause dramatic problems with phosphorous-related eutrophication. However, this may also indicate that the simplifications in the spatial modeling of phosphorous emissions (in particular distance to water bodies) are inadequate and lead to overestimation of the emitted phosphorous. The newly available map of field connection to surface waters ("Gewässeranschlusskarte" [169]) may provide the opportunity to achieve a more correct estimation of phosphorous reaching surface waters at a regional scale.

Although the estimate of total Swiss agricultural phosphorous emissions used in the above comparison comprises very high uncertainties, the comparison indicates that the absolute amount of phosphorous emitted in the scenarios assessed here should be considered with a lot of precaution. However, the relative comparison of the eutrophication potential between scenarios can be considered more reliable, since all these scenarios were modeled with the same assumptions and therefore a similar bias.

5.6.3 Regionalization of groundwater use effect factor

The effect factor used for the estimation of potential terrestrial species loss due to groundwater consumption [36] was developed for the Netherlands, although the plant species it considered are considered representative for temperate regions in Europe and thus also adequate for the Swiss lowlands. The method uses a species sensitivity distribution to soil moisture based on Ellenberg indicator values [170] (values which indicate the typical occurrence ranges of individual species to several environmental parameters such as soil moisture). A similar effect factor could also be calculated specifically for Switzerland, using the Landolt indicator values [171] (an equivalent of the Ellenberg indicator values developed specifically for Swiss flora), thus reducing the uncertainty in the result related to using a non-regionalized effect factor (this was however beyond the scope of this thesis and was not done due to time constraints).

Note that the estimation of terrestrial biodiversity loss due to groundwater extraction (Van Zelm et al. 2011) does not use the same method as the estimation of reduction in potential terrestrial biodiversity on farmland due to changes in field management (SALCA Biodiversity). The method by Van Zelm et al. (2011) estimates the reduction of potentially occurring terrestrial plant biodiversity in natural ecosystems, due to a drop in groundwater table (the vegetation types considered are nutrient-poor grassland (low herbaceous vegetation), pine forest, spruce forest, deciduous forest, and heath [36]). The method in SALCA Biodiversity estimates the reduction of potentially occurring species of cropland and grassland flora, insects, birds, amphibians, arachnids, small mammals, and mollusks on agricultural land, due to changes in field management (crop choice, fertilizer management and tillage in the present case) [129]. Thus these two indicators consider principally different partitions of terrestrial biodiversity and different impact pathways.

5.6.4 Limitations

Uncertainties, concerns about extrapolation to other regions, and about consequences of decreases in productivity on a broader spatial scale are all similar in the regional scale assessment as in the farm scale assessment (as discussed in Chapter 4.4.5). In addition for the regional scale, the regional agricultural system was simplified in order to allow a compromise between spatial explicitness and computational feasibility (see Chapter 5.2.1). This simplification causes additional uncertainties in the results, although it is estimated that these would not affect the relative importance of impacts between the different scenarios (with the possible exception of internal transportation). Indeed, the parameters identified in the emissions modeling as highly sensitive to location were kept spatially explicit. Spatial detail within the case study region for other parameters is expected to allow only a marginal improvement in precision compared to the uncertainties residing in the emissions models themselves as well as in the impact assessment models for each impact category. Internal transportation (for example of manure or fodder) was not modeled here (as explained in Chapter 5.2.1): this may however contribute in a non-negligible way to further global warming potential for example.

The regional adaptation strategies did not consider any scenarios of conversion between land use types (e.g. reduction in total agricultural land use area) in the case study region, although this may also change by 2050. Such a change would affect the total production in the region, but may not necessarily affect the impacts per ha or per MJ dig. en. of the remaining agricultural production (as shown by a similar study in the Greifensee case study region [172]).

6 CRITICAL APPRAISAL OF THE THESIS AND DISCUSSION

6.1.1 Impacts related to irrigation and aquatic biodiversity

The LCIA method developed in Chapter 2 addresses quantitative water consumption. However, water quality should also be assessed when addressing impacts on aquatic ecosystems. Water quality issues which were considered in this thesis are the existing LCA impact categories aquatic ecotoxicity and aquatic eutrophication, as well as the issue of water temperature. Further issues such as effects of agricultural practice on erosion and subsequent river sediment load were not considered.

Further potential impacts related to agricultural water consumption for irrigation include impacts on riparian ecosystems, lakes, groundwater ecosystems, wetlands and on highly water-dependent terrestrial taxa (e.g. water birds). In the more critical case study region considered in this thesis, the river ecosystem had a relatively rich biodiversity and importance, whereas riparian ecosystems and wetlands connected to the river or groundwater were hardly present, and therefore were not specifically addressed. Agricultural water abstraction from lakes near the case study region does occur, however the level of these lakes is artificially regulated, thus shifting the effects of lake water abstraction back to the river system at the outflow of the lakes. The impacts of groundwater abstraction on terrestrial ecosystems was assessed in Chapter 5.5.1, however impacts on the groundwater ecosystem itself were not addressed. Indeed groundwater ecosystems have yet to be studied to an extent enabling such an impact assessment [173]. Highly water-dependent terrestrial taxa were not considered here. Nevertheless, a disturbance of aquatic ecosystems may also be expected to affect such taxa too [84], via changes in habitat and food supply for example.

This thesis used two methods allowing an assessment of the impacts of changes in river discharge on aquatic biodiversity. The first, relying on species-discharge relationships, presents the advantage of being more generic and applicable at global coverage with relatively low data requirements; the latter is also an advantage when addressing future assessments where data predictions may be limited. In this sense, it provides a useful screening tool allowing the identification of problematic situations. However, it also entails high uncertainties, in particular for rivers with low discharges. Indeed, the shape of the species-discharge relationships used here is largely driven by a small number of data points in the range of larger discharges, whereas the data points at low discharges show a high variability. This variability may be caused by a shift in the relative importance of different drivers of aquatic biodiversity at

different scales (although changes in discharge may have a significant effect at certain scales, they may be overridden by changes in other parameters such as channel morphology at other scales [174]). Additionally, this method will not reveal site-specific particularities (although this is not an objective of LCA and therefore does not restrict applicability in LCA, it may restrict applicability within a local planning context) and does not rely on a proven causal mechanism. It is also not yet available at a global scale, for which characterization factors would have to be calculated. On the other hand, the impact assessment method relying on a water temperature model presents the benefit of being site-specific, with a flexible scale of application from river reach to watershed. It relies on a mechanistic impact pathway, and allows a higher spatial resolution (temperature changes and their impacts can be evaluated for any number of sites in a watershed) and a temporally-explicit assessment of changes in temperature and their potential impact on aquatic biodiversity. However, it is much more effort-consuming to apply for broader spatial coverage, principally because of a higher data requirement. Integrating both models in their present form is not directly possible since the variables they consider are not independent (since they both rely on changes in discharge, either directly or indirectly, the estimated loss of species from both methods cannot be directly added since this could involve double counting). However, the use of multivariate models (which consider multiple drivers and their interactions) for species richness and biodiversity metrics prediction in aquatic ecosystems would be an ideal approach in order to integrate all impact pathways related to water [78, 174, 175]. Existing efforts in this direction are typically data-demanding and tend to be site-specific: however, they may prove to have a high potential if adapted for application in an LCA context.

Temporal patterns and seasonality in discharge and thermal regime disturbance are expected to play a significant role on the actual impacts on aquatic species: parameters such as timing of low flow [176] and timing of life-stage-specific temperature tolerance threshold excess [70] have been found to have a significant effect on the viability and richness of aquatic populations. Ideally, LCIA methods should include sufficient temporal resolution to reflect such variations in ecosystem vulnerability. A candidate method for application in LCA is contained in the temperature effect model used here, and compares daily maximum and minimum temperatures with species life-stage-specific tolerance thresholds. This could not be used in the current study of future scenarios (as discussed in Chapter 5.4.2), since simulations of daily minimum and maximum temperatures were not available. Nevertheless, this approach would be of interest for application in studies where such data is available. The species-discharge relationship can also be tested for further parameters other than yearly average discharge, based on a selection of parameters and a method recommended in a previous study [167] (such as

timing of low flows, frequency of low flows etc.). This might reveal additional relationships between species richness and other hydrological parameters, which could improve the sensitivity of predicted impacts to temporally-explicit interventions in the hydrological regime for example.

This thesis used species richness as an indicator of biodiversity, due to limitations in data availability for other indicators (such as abundance, genetic diversity, structural and functional indicators etc., see Curran et al. 2011 [44]). This is however not sufficient to illustrate all the aspects related to ecosystem viability, health and service provision. Assessments of biodiversity in LCA should ideally also include other indicators of ecosystem quality [44, 174, 177], where data is available (e.g. species abundance, ecosystem functionality, genetic diversity as well as further taxa). Further efforts in data collection for aquatic ecosystems is urgently needed [78, 178], in particular for developing countries [28]: indeed, the availability of species occurrence, species richness and species abundance data is necessary for assessing the potential impacts of water use on aquatic ecosystems, but is available mainly for developed countries, and even there datasets have large gaps.

6.1.2 Assessment of agricultural adaptation strategies to climate change

The implications of agricultural adaptation strategies at a higher spatial scale (such as at the national and international scale) were not addressed in this thesis. Indeed, the scope of this thesis focuses on farm adaptation, and the regional impacts that result from the accumulation of farm impacts in a region as well as their mitigation strategies within a region. Nevertheless, some of the strategies imply changes in local productivity, which may have broader consequences on the national and ultimately international food production system (due to changes in imports and exports for example). These should ideally also be considered when deciding which adaptation strategies should be favoured, for example using a consequential LCA approach [179]. Recent research has also recommended that an LCA of a region for the support of regional land use policy planning should consider not only the impacts of the local production, but also the impacts of the local consumption (in the sense that the impacts caused by the local consumption requirements should also be considered, whether they occur in the region or not); benefits (such as ecosystem services, e.g. provision of clean air and water) should be considered alongside impacts, in order to reflect the "total responsibility" of a region [152]. This thesis focused only on the assessment of the impacts of production in a region. However, the potential consequential effect chain of a decrease in production in the region is discussed here in a qualitative way (as recommended if a quantitative analysis is not feasible [179]; due to extreme uncertainties related to global production in 2050, a quantitative analysis is not considered feasible within the scope of this thesis; also, most

available quantitative global market models would not capture such a marginal change in Swiss imports [180] and are built at a more aggregated scale). The preservation regional scenario for the Broye implies a significant decrease in production of wheat, potatoes and sugar beet (representing respectively 36%, 32% and 25% of the total loss of 5.5 bio MJ dig. en. compared to the reference scenario), whereas the productivity scenario implies a decrease in rapeseed production despite an overall increase in the production of MJ dig. en. This decrease in local production is assumed to correspond to an increase in imports of products from elsewhere, since of the four possible reactions [179], an increase in productivity within the region, or an increase in agricultural land use area within the region, is excluded per definition by the scenario, and it is assumed that Switzerland will have the financial capacity to buy the missing products rather than renounce their use [180]. (The option that corresponding exports of the products are simply decreased is also disregarded, since Switzerland is already largely a net importer of wheat products and other cereals, potatoes, sugar products and rapeseed oil [181]). According to the FAO [182], in 2050, developed countries will be net exporters of cereal crops, whereas developing countries will be net exporters of oil crops and sugar products. Thus if it is assumed that Switzerland were to apply policies towards the application of the preservation scenario throughout the whole country, the required increase in production elsewhere would ultimately be provided from developed countries (in the case of cereals) and developing countries (in the case of oil and sugar products). (Although direct imports into Switzerland may come from other regions, the whole displacement chain can be roughly considered to end at the net exporters). Major exporters for cereals in 2050 are projected to remain the USA, Russia, Eastern Europe and to a lesser extent the EU-27. The dominant exporter for sugar products is Brazil, whereas major exporters of oil products (no distinction of crop) are Malaysia, Indonesia, Argentina, Brazil and Paraguay (although this principally concerns palm oil). No projections are made for potatoes by the FAO. The production increase required in such exporting countries is expected to result mainly from increases in yields for wheat in particular (99% from yield increase compared to 1% from harvest area expansion, which includes both land expansion and cropping intensity increase). For all crops together, yield increase is expected to contribute to 80% of production increases in 2050, although land use expansion is expected to contribute a larger than average share to production increases for oil crops. The yield increases (all crops together) are expected to be provided mainly by increases in fertilization intensity (whereas irrigation water withdrawals are expected to increase only by 6%, the increase in water demand being offset in a large part by an increase in irrigation efficiency). Thus the preservation scenario assessed here may ultimately cause impacts related to more intensively fertilized wheat production in developed countries, as well as more intensively fertilized sugar beet or sugar cane production in developing countries. The productivity

scenario may imply impacts related to more intensive oil crop production and/or expansion of land use for oil crop production in developing countries. This does not include potential differences in impacts caused by changes in production by intermediate importers and exporters. The dominant source countries of direct imports to Switzerland over the last 10 years, according to the FAO statistics [183], consist of: Germany, France and Canada for wheat, and Germany for wheat flour; Germany for sugar beet, and Germany, France and to a lesser extent Brazil and other South American countries for refined sugar; Netherlands, Israel, France, Germany for potatoes; and France and Germany for rapeseed and rapeseed oil. According to the FAO Outlook database [184], the EU-27 will maintain its exports and increase its production of wheat and oilseeds until the horizon 2021, whereas its exports and production of sugar and sugar beets will decrease. Canada will also maintain its exports and increase its production of wheat. Brazil will increase its exports and production of sugar. Potatoes are not included in the outlook.

In summary, it can be assumed that at least until 2021, Switzerland will be able to import its additional missing wheat and rapeseed products from the EU-27 and Canada as previously, but will however probably have to shift more of its sugar sourcing to Brazil for example. Until 2050, wheat imports may also be increasingly sourced from eastern European countries. According to average inventories available in the ecoinvent database [141], per kg product and according to the impact categories used in this thesis (see Table 8), wheat production in Germany has lower impacts than wheat production in Switzerland for most impact categories (except global warming potential and non-renewable energy use). Wheat production in France has higher impacts for roughly half of the indicators (only the toxicity indicators as well as marine eutrophication and ozone formation are lower). Wheat production in the USA (as a proxy for Canada) has mostly higher impacts (except for marine eutrophication and terrestrial ecotoxicity). Rapeseed production shows higher impacts in France, and lower impacts in Germany (except for ozone formation and human toxicity), compared to Swiss rapeseed production. The impacts of sugar production in Brazil are variably higher or lower than those of sugar production in Switzerland according to the impact category. Although this comparison is associated with high uncertainties (due to the many generalizing assumptions made above and in the source data), this would suggest that a slight shift of wheat and rapeseed provisioning to the above-mentioned exporting countries could cause an increase in the overall production impacts if the main source is Canada (for wheat), a decrease in impacts if the main source is Germany, and a variable (for wheat) to increasing (for rapeseed) trend if the main source is France. The shift of sugar provisioning to exporting countries may cause variable trends in impacts in 2050 (although transport impacts would also have to be accounted for. These however generally form a small share of the overall impacts of agricultural products which do not

require air transport or energy-intensive storage conditions, which is the case of the crops addressed here).

A full sustainability assessment should involve not only economic and environmental criteria, but also social criteria [185]. Again this was not the focus of this thesis. Consideration of such criteria would contribute a further perspective on the question of sustainable agricultural adaptation to climate change. From an LCA perspective, the use of economic criteria should also be completed and formalized in order to enhance compatibility with the environmental criteria [186].

The scenarios assessed in this thesis represent only a part of potential agricultural adaptation and watershed management options. Other potential options include modification of land use shares between agriculture and other uses (such as forestry), use of different crop varieties (including genetically engineered crops) and new crop types (such as sorghum and other drought-tolerant crops) and river restoration, which would merit further research as to their potential effects. These were however not include in the AGWAM project nor in this thesis, since the focus was on assessing the potential of existing management options for agricultural adaptation to climate change.

In general, water management is recommended to be operationalized at the watershed scale [48]. Integrated watershed management should not only consider multiple stressors on water resources and ecosystems, but should also consider the needs of other water users in order to optimize water management. Similarly, planning for sustainable agriculture should occur at the landscape scale, rather than only at the farm scale: this enables simultaneous management of all risks related to climate change and agricultural production, by including all the associated factors interlinked in a complex system [187]. An example of an integrated assessment and optimization model, using genetic optimization and multi-agent system modeling, has been developed and tested for a case study in the USA, and represents an interesting approach for case studies where the focus is set on impacts on aquatic biodiversity [188]. Such an approach also enables consideration of the economic cost of potential management options, which is necessary for the final selection of management and mitigation option strategies by policy-makers. The consideration of stakeholder acceptance of mitigation strategies is also essential in order to ensure their final realization. Indeed the confrontation of the strategies assessed here with local stakeholders from the case study region (see Chapter 4.4.8) showed that acceptance may differ from the optimum proposed by the models: stakeholders expressed preferences based on criteria that were

not included in the models, such as tradition, availability of know-how, flexibility in working hours, and existence of market infrastructure and demand for the products.

LCA enabled the consideration of multiple potential environmental impacts, in order to ensure that relevant impacts were not neglected. However, the application of LCA to numerous future scenarios (where many potential options are likely to be addressed) is time-consuming. The work flow of the tools used in this thesis was automated in order to reduce manual time requirements and minimize human error, and proved to be a useful approach for this context. The standard inventories for farm LCA are quite detailed, however many of the inventory flows listed were not considered in the models producing the agricultural adaptation scenarios. Simplification of these inventories was justified in this case by the high uncertainties related to future agricultural practices, and the use of a limited amount of adaptation variables in the scenarios, that focus on large shifts in agricultural practice (rather than fine-scale tuning of individual inventory flows). A partial sensitivity analysis was conducted in order to identify the inventory flows which dominate impact outcomes; however a systematic sensitivity analysis would be helpful in order to identify which inventory flows require more detailed attention in other contexts (for example, in environmental optimization of real farm management using LCA, where detailed inventory flows are also available and may be meaningful).

Applying PCA to the LCA results was useful in identifying the relevant environmental trade-offs. However, it presents two limitations: (1) it is only applicable where the number of cases (or scenarios) assessed is larger than the number of variables (or impact categories). If this is not the case, a correlation analysis allows an alternative assessment of trade-offs; (2) it informs on the variation of impacts between the scenarios, but not on the absolute magnitude of impacts. Thus in parallel to such an approach, the assessment of the absolute magnitude of each impact category is nevertheless recommended, in order to identify if any impact category is of particular concern in absolute terms (which may not correspond to its importance in the PCA).

The farm and regional scale optimization models both use different goals and different optimization approaches. This means that their outcomes are not entirely comparable even if they are scaled up or down to a comparable level. However, global warming potential and aquatic biodiversity loss were compared at both scales, and it is interesting to note that they are within similar orders of magnitude: global warming potential ranges from 1910 to 6533 kg CO_2 eq. /ha*y at the farm scale and from 3818 to 5099 kg CO_2 eq. /ha*y at the regional scale; aquatic biodiversity loss reaches a maximum of 5E-10

GSE*y/ha*y at the farm scale and 8E-11 GSE*y/ha*y at the regional scale (this is lower because it includes farms in the hill areas which require less irrigation than the modeled farm in the lowland area). Consistent modeling at both scales would present the advantage of consistent interpretations and allow an analysis of the effect of scale; however it is expected that regional assessments will continue to require certain simplifications compared to an aggregation of farm units [189]. Nevertheless, the observation of trends and drivers at the farm scale can give important indications to policy-makers at the regional scale, by enabling identification of effective incentives for achieving particular goals (for example water quotas or changes in subsidies for reducing aquatic biodiversity loss). Conversely, the observation of aggregated impacts occurring at a regional scale provide indications of priority issues (such as aquatic biodiversity or freshwater eutrophication) that must be addressed at the farm management level.

Several common trends can be observed at both assessment scales:

- Water use and its impacts are a major environmental trade-off in adaptation to climate change
- Intensification tends to lead to higher absolute impacts per area and higher productivity, but generally lower impacts relative to production (except concerning aquatic biodiversity loss), whereas extensification tends to lead to lower absolute impacts and lower productivity, often resulting in higher impacts relative to production.

Both these points are in agreement with observations made in a recent study of food production in the UK and abroad [189], which similarly found that impacts tend to be lower for regions with higher productivity, and that impacts of water use may present a major trade-off in production systems which minimize global warming potential. Terrestrial biodiversity has generally been estimated to be lower on intensive agricultural land than on extensive agricultural land [190, 191]. Various opinions exist on whether extensification of existing agricultural land and extension of agricultural land use is preferable to intensification of existing agricultural land, in order to meet increased food demands while mitigating impacts on terrestrial biodiversity. Intensification of low-intensity existing agricultural land may be preferable to extension of agricultural land use in order to increase productivity with minimal biodiversity impacts [192] (as confirmed by the decrease in reduction of potential terrestrial biodiversity per MJ dig. en. for the productivity scenario shown in Figure 25). However, the most efficient gain in biodiversity may be achieved through further extensification of low-intensity agricultural land in other cases [192] (as confirmed by the decrease in reduction of potential terrestrial biodiversity per ha*y for the preservation scenario shown in Figure 26). A third perspective states that further intensification of

existing agricultural land may even lead to a decrease in total agricultural land use, but also to decreases in biodiversity on agricultural land which are not fully compensated by increases in biodiversity through development of natural ecosystems on the abandoned land [191]. The results of Chapter 5.3.2 suggest that intensification would reduce impacts on biodiversity when related to productivity: this indicates that MJ dig. en. would be produced in a more efficient way for biodiversity through intensification than through extensification. However, as mentioned above, further MJ dig. en. could be produced on additional land converted to agriculture: the overall impact on biodiversity would then also depend on the loss of biodiversity occurring when converting land from the previous land use to extensive agriculture.

7 ACHIEVEMENTS, CONCLUSIONS AND OUTLOOK

This thesis combined the development of a life cycle impact assessment method for river water consumption with an extensive LCA case study at two spatial scales, relevant for farm management decisions and regional policy decisions respectively. The new life cycle impact assessment method fills a gap in the range of available methods, which is important in order to ensure LCA achieves the goal of a comprehensive assessment including all relevant impacts. Additional impact assessment approaches providing more detail for the local context were applied in order to evaluate specific water management options at the regional scale. Together, this provides a support for policy-makers as well as farmers towards mitigating the environmental impacts of agricultural adaptation to climate change.

The characterization factor developed in Chapter 2, relating river water withdrawals to impacts on aquatic biodiversity, considers the location of withdrawal within a river basin, includes fish and macro-invertebrate taxa, and uses a novel measure of impacts on biodiversity in the form of global species extinction equivalents. The use of aquatic biodiversity loss rather than just m^3 of water consumed (such as in water footprinting [193]) allows a weighting of the amount of water used by the potential impact this would have on ecosystems, contributing relevant additional information differentiating the vulnerability of ecosystems affected, and providing a unit with an absolute reference (global species extinction). Compared to previous efforts, spatial resolution was increased based on ecologically relevant spatial units, additional taxa were included and sensitivity of the approach to choice of regression model and biogeographic region was assessed. This approach should be applicable for broad spatial coverage around the globe, and can be integrated in an LCA framework. However, it presents two main limitations: it relies on a statistical relationship for which a causal relationship is still to be verified, and it is a generic method: it does not present sufficient detail for in-depth site-specific studies. Thus it can be recommended for LCAs at national and international scales, but is not an optimal approach for local studies (as is often the case in LCIA, for example for ecotoxicity).

This thesis also tested an alternative approach, relating river water withdrawals to changes in water temperature, in turn linked to potential species loss. This approach is applicable at smaller scales (such as sub-catchments) for a more detailed assessment of local impacts, and provides a potential causal relationship. However, as the case study showed, it is not always a relevant impact pathway: although changes in water temperature due to climate change were significant, further effects of irrigation

withdrawal on water temperature were negligible, and water temperature changes did not lead to extended impacts on aquatic fauna. Depending on the site under study, other pathways may therefore have more priority when seeking to address and mitigate impacts of river water use and management of rivers on aquatic biodiversity (for example eutrophication or toxicity). Including multiple impact pathways in a multivariate model is challenging and data-intensive, and outcomes tend to be highly site-specific. Thus although it may provide more detailed results for local case studies, such an approach is as yet of limited applicability in an LCA framework; simplifications would be required, which in turn would affect the uncertainty involved. The combination of different bottom-up approaches, each modeling the impacts on biodiversity of a different stressor through an independent impact pathway, could also be a solution; however impact pathways are often inter-related and therefore aggregation could lead to double-counting of impacts. For example, the simultaneous consideration of both the approaches used in this thesis is not directly feasible, since the variables they consider are not independent, and therefore the estimated impacts cannot be simply aggregated without double-counting.

Major results from the thesis as a whole lead to the following conclusions: at the farm scale, the environmental impacts of Swiss agricultural production (related to the amount produced) are expected to increase in the future climate. Strategies maximizing farm economic profitability in future aggravate water-related impacts, however most other environmental impacts (per amount produced) are lower for economically optimized farms than for farms without adaptation to the future climate, although in future, productivity and eco-efficiency decrease. Water quotas or pricing could avoid most impacts on aquatic biodiversity caused by irrigation withdrawals, however at the cost of productivity and other environmental impacts. Assuming economic farm optimization represents farmers' adaptation behaviour in the absence of other incentives, this suggests that additional policies and incentives are required if productivity is to be maintained and environmental impacts mitigated. Changes in environmental impacts are affected by changes in agricultural product prices as much as by climate change; on the other hand, farm subsidy changes (as currently planned) have a negligible effect in most cases. Different farm types and different regions may not always show similar responses and sensitivity to changes in climate, prices and subsidies (as shown for two farm types and two regions in this thesis). Therefore if particular goals are desired concerning environmental impacts or productivity, policies may need to intervene at a sub-national spatial scale, with specific actions targeting different farm types.

At the regional scale, strategies maximizing productivity are able to increase productivity and eco-efficiency, but at the cost of aquatic biodiversity. Thus the major environmental trade-off in agricultural adaptation to climate change was found to be between impacts on aquatic biodiversity and most other impact categories (including global warming potential, resources use, toxicity etc.). The impacts on aquatic biodiversity related to irrigation water use can therefore be considered as a relevant component of environmental impacts in this case, and should be addressed in similar contexts of climate change, even for regions where screening methods such as the WSI show a low water stress. Higher productivity generally leads to higher total impacts within the region, but lower impacts relative to the amount produced, for impact categories other than aquatic biodiversity loss. In this sense, global warming potential was generally mitigated through adaptation to climate change.

At the regional scale, aggregated impacts which potentially reach levels of concern include aquatic biodiversity loss (potential loss of up to one fifth of species in the watershed due to climate change and irrigation) and freshwater eutrophication (emissions of nutrients up to ten times the national average). Therefore regional impact management in the case study region should focus on mitigating these impact categories. Two approaches for mitigation of impacts on aquatic biodiversity were investigated: the use of groundwater for irrigation and the increase of riparian shading along the river. Both were found to achieve a partial mitigation of the impacts on the river biodiversity in the case study region, however this is not sufficient to mitigate the entire impacts of climate change and irrigation water requirements on the river ecosystem. Additionally, certain policy measures assessed were effective in abating impacts on aquatic biodiversity, such as limiting the water withdrawal allowance by quotas, or reorienting the production in the region to grasslands and animal production. These however imply a decrease in productivity. Therefore further management measures (such as use of new crops and crop varieties, irrigation water sourcing from lakes, river restoration) may be necessary in order to compensate the impacts to a fuller extent without compromising the productivity of the region for human nutrition.

Following from the above conclusions, several recommendations for further research can be made:

- This thesis provided a characterization factor for the case study of interest, using the LCIA method for river water consumption developed here. The extensive application of this method to other regions in order to provide a set of characterization factors with broad spatial coverage would greatly enhance the operational applicability of the method within LCAs. The novel unit for the

assessment of biodiversity loss used in this thesis can be applied to other LCA impact categories too, in order to enhance comparability between impacts. This would require the adaptation of existing impact category models and characterization factors.

- The two approaches used in this thesis to address the impacts of river water consumption used a generic impact pathway and a single-driver impact pathway respectively. The integration of multiple impact pathways affecting aquatic biodiversity (such as eutrophication, toxic contamination, water consumption, and changes in the thermal regime) in a multivariate model would enable the consideration of impact interaction and integrate impacts in a consistent way, while providing more holistic models for the prediction of aquatic biodiversity loss, capable of reflecting limiting conditions for specific sites. Such a model would also allow investigation of further management options, where impacts through one pathway may be compensated by management intervening on another impact pathway, thus providing more flexibility to decision-makers. It is however expected that such models would be associated with high levels of uncertainty and high data requirements for broad spatial coverage, which may limit their feasibility for large-scale application in LCA.

- The consideration of temporal variations in flow and thermal regime and of extreme values rather than just yearly average discharge and daily average temperature, and relation to specific tolerances of species according to their life stages, would allow identification of acute stress and assessment of its impacts. This would complement the assessment of chronic stress measured with the average values, and enable a more detailed assessment of management options targeting temporal variables.

- This thesis focused on assessing the environmental impacts of agricultural adaptation scenarios to climate change; however a full sustainability assessment should also include relevant social and economic criteria. The latter were partly considered in the farm scenarios used here (profit and variability of profit), however social criteria were not included. These could include work load for the farmers, but also social impacts related to required inputs and imports associated with the scenarios.

- The assessments conducted here focused on farm and regional scale management. However the scenarios assessed imply changes in productivity, which imply further changes in other components of the system not addressed in detail here, such as changes in import and export amounts, or changes in consumption requirements. Consideration of such effects using national and international market models and consequential LCA for example would provide a more exhaustive perspective on the overall environmental implications of the scenarios, reaching beyond the local implications, and supporting strategy formulation at a national level.

REFERENCES

1. OcCC; ProClim. *Climate change and Switzerland 2050*: Bern, Switzerland, 2007.
2. Smit, B.; Skinner, M.W. Adaptation options in agriculture to climate change: a typology. *Mitig Adapt Strat Glob Change* **2002**, *7*, 85-114.
3. Howden, S. M.; Soussana, J.-F.; Tubiello, F.N.; Chhetri, N.; Dunlop, M.; Meinke, H. Adapting agriculture to climate change. *PNAS* **2007**, *104* (50), 19691-19696.
4. Kirchmann, H.; Thorvaldsson, G. Challenging targets for future agriculture. *European Journal of Agronomy* **2000**, *12*, 145-161.
5. Robertson, G.P.; Swinton, S.M. Reconciling agricultural productivity and environmental integrity: a grand challenge for agriculture. *Front Ecol Environ* **2005**, *3* (1), 38-46.
6. Fuhrer, J.; Lehmann, B.; Holzkämper, A.; Calanca, P.; Gaillard, G; Hellweg, S. *Water demand in Swiss agriculture, and sustainable adaptive options for land and water management to mitigate imapcts of climate change (AGWAM) - Full proposal*; National Research Programmes: Bern, Switzerland, 2009.
7. Lehmann, N. *How climate change impacts on local cropping systems: a bioeconomic simulation study for western Switzerland*. PhD Thesis, ETHZ, Zürich, Switzerland, 2013.
8. Klein, T.; Holzkämper, A.; Calanca, P.; Fuhrer, J., In *Identifying optimum strategies for land management adaptation to climate change - a multiobjective approach*, Proceedings of International Environmental Modelling and Software Society, Leipzig, 2012; Seppelt, R.; Voinov, A.A.; Lange, S.; Bankamp, D., Eds.
9. Klein, T. *Adapting agricultural land management to climate change – A regional multi-objective optimization study*. PhD thesis, University of Bern, Bern, Switzerland, 2013.
10. Van Der Werf, H. M. G.; Petit, J. Evaluation of the environmental impact of agriculture at the farm level: a comparison and analysis of 12 indicator-based methods. *Agriculture Ecosystems & Environment* **2002**, *93*, 131-145.
11. Thomassen, M.A.; De Boer, I. J. M. Evaluation of indicators to assess the environmental impact of dairy production systems. *Agriculture, Ecosystems & Environment* **2005**, *111*, 185-199.
12. Nemecek, T.; Freiermuth-Knuchel, R.; Alig, M.; Gaillard, G., In *The advantages of generic LCA tools for agriculture: examples SALCAcrop and SALCAfarm*, Proceedings of 7th Int. Conf. on LCA in the Agri-Food Sector, Bari, Italy, 2010; Notarnicola, B., Ed. pp 433-438.
13. Küstermann, B.; Christen, O.; Hülsbergen, K-J. Modelling nitrogen cycles of farming systems as basis of site- and farm-specific nitrogen management. *Agriculture, Ecosystems & Environment* **2010**, *135*, 70-80.
14. Häni, F.; Braga, F.; Stämpfli, A.; Keller, T.; Fischer, M.; Porsche, H. RISE, a tool for holistic sustainability assessment at the farm level. *International Food and Agribusiness Management Review* **2003**, *6* (4), 78-90.
15. Eckert, H.; Breitschuh, G.; Sauerbeck, D.R. Criteria and standards for sustainable agriculture. *J. Plant Nutr. Soil Sci.* **2000**, *163*, 337-351.
16. Van Der Werf, H. M. G.; Tzilivakis, J.; Lewis, K.; Basset-Mens, C. Environmental impacts of farm scenarios according to five assessment methods. *Agriculture, Ecosystems & Environment* **2007**, *118*, 327-338.
17. Payraudeau, S.; van der Werf, H.M.G. Environmental impact assessment for a farming region: a review of methods. *Agriculture, Ecosystems & Environment* **2005**, *107*, 1-19.
18. Acosta-Alba, I.; Lopez-Ridaura, S.; van der Werf, H. M. G.; Leterme, P.; Corson, M. S. Exploring sustainable farming scenarios at a regional scale: an application to dairy farms in Brittany. *Journal of Cleaner Production* **2012**, *28*, 160-167.

19. Nemecek, T.; Dubois, D.; Huguenin, O.; Gaillard, G. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. *Agricultural Systems* **2011**, *104*, 217-232.
20. Mouron, P.; Nemecek, T.; Scholz, R.W.; Weber, O. Management influence on environmental impacts in an apple production system on Swiss fruit farms: combining life cycle assessment with statistical risk assessment. *Agriculture Ecosystems & Environment* **2006**, *114*, 311-322.
21. Nemecek, T.; Huguenin, O.; Dubois, D.; Gaillard, G; Schaller, B.; Chervet, A. Life Cycle assessment of Swiss farming systems: II. Extensive and intensive production. *Agricultural Systems* **2011**, *104*, 233-245.
22. Hersener, J.-L.; Baumgartner, D.; Dux, D.; Aeschbacher, U.; Alig, M.; Blaser, S.; Gaillard, G.; Glodé, M.; Jan, P.; Jenni, M.; Mieleitner, J.; Müller, G.; Nemecek, T.; Rötheli, E.; Schmid, D. *Zentrale Auswertung von Ökobilanzen landwirtschaftlicher Betriebe (ZA-ÖB).* Forschungsanstalt Agroscope Reckenholz-Tänikon ART: Zürich/Ettenhausen, Switzerland, 2011.
23. IPCC. *IPCC Fourth Assessment Report: Climate Change 2007*: Cambridge, UK and New York, USA, 2007.
24. (EEA), European Environment Agency. *Water resources accross Europe - confronting water scarcity and drought*: Copenhagen, 2009; p 55.
25. ProClim, OcCC -. *Climate change and Switzerland 2050*: Bern, 2007.
26. Goedkoop, M.; Heijungs, R.; Huijbregts, M. A. J.; De Schryver, A.; Struijs, J.; van Zelm, R. *ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level.*; 1; Pré Consultants, CML Leiden University, Radboud University, RIVM: Netherlands, 2009.
27. Fuhrer, J. *Bewässerungsbedarf und Wasserdargebot unter heutigen und künftigen Klimabedingungen*; Agroscope Reckenholz-Tänikon research station ART: Zürich, Switzerland, 2012.
28. Malmqvist, B.; Rundle, S. Threats to the running water ecosystems of the world. *Environmental Conservation* **2002**, *29* (2), 134-153.
29. *The United Nations World Water Development Report 3: Water in a changing world*; World Water Assessment Program: Paris and London, 2009.
30. Richter, B.D.; Mathews, R.; Harrison, D.L.; Wigington, R. Ecologically sustainable water management: managing river flows for ecological integrity. *Ecological Applications* **2003**, *13* (1), 206-224.
31. Rack, M.; Valdivia, S.; Sonnemann, G. Life Cycle Impact Assessment - where we are, trends, and next steps: a late report from a UNEP/SETA Life Cycle Initiative workshop and a few updates from recent developments. *Int J LCA* **2013**; DOI 10.1007/s11367-013-0569-1.
32. Kounina, A.; Margni, M.; Bayart, J.-B.; Boulay, A.-M.; Berger, M.; Bulle, C.; Frischknecht, R.; Koehler, A.; Mila i Canals, L.; Motoshita, M.; Núñez, M.; Peters, G.; Pfister, S.; Ridoutt, B. G.; Van Zelm, R.; Verones, F.; Humbert, S. Review of methods addressing freshwater use in life cycle inventory and impact assessment. *Int J LCA* **2012**, *Online*.
33. Berger, M.; Finkbeiner, M. Methodological Challenges in Volumetric and Impact-Oriented Water Footprints. *Journal of Industrial Ecology* **2013**, *17* (1), 79-89.
34. ISO, Environmental Management - Life Cycle Assessment - Requirements and guidelines. In International Organisation for Standardisation: Geneva, Switzerland, 2006b.
35. Pfister, S.; Koehler, A.; Hellweg, S. Assessing the Environmental Impacts of Freshwater Consumption in LCA. *Environ. Sci. Technol.* **2009**, *43* (11), 4098-4104.
36. van Zelm, R.; Schipper, A. M.; Rombouts, M.; Snepvangers, J.; Huijbregts, M. A. J. Implementing groundwater extraction in life cycle assessment: characterization factors based on plant species richness. *Environ. Sci. Technol.* **2011**, *45* (2), 629-635.

37. Mila i Canals, L.; Chenoweth, J.; Chapagain, A.; Orr, S.; Anton, A.; Clift, R. Assessing freshwater use impacts in LCA: Part I-inventory modelling and characterisation factors for the main impact pathways. *International Journal of Life Cycle Assessment* **2009**, *14* (1), 28-42.
38. Hanafiah, M.M.; Xenopoulos, M.A.; Pfister, S.; Leuven, R.S.E.W.; Huijbregts, M. A. J. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. *Environ. Sci. Technol.* **2011**, *45* (12), 5272-5278.
39. Xenopoulos, M.A.; Lodge, D.M.; Alcamo, J.; Märker, M.; Schulze, K.; van Vuuren, D.P. Scenarios of freshwater fish extinctions from climate change and water withdrawal. *Global Change Biology* **2005**, *11*, 1557-1564.
40. Spooner, D.E.; Xenopoulos, M.A.; Schneider, C.; Woolnough, D.A. Coextirpation of host-affiliate relationships in rivers: the role of climate change, water withdrawal, and host-specificity. *Global Change Biology* **2011**, *17*, 1720-1732.
41. Arthington, A.H.; Bunn, S.E.; Poff, N.L.; Naiman, R.J. The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* **2006**, *16* (4), 1311-1318.
42. McGarvey, D.J.; Ward, G.M. Scale dependance in the species-discharge relationship for fishes of the southeastern USA. *Freshwater Biology* **2008**, *53*, 2206-2219.
43. Poff, N.L.; Zimmerman, J.K.H. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology* **2010**, *55*, 194-205.
44. Curran, M.; De Baan, L.; De Schryver, A.; van Zelm, R.; Hellweg, S.; Koellner, T.; Sonneman, G.; Huijbregts, M. A. J. Toward meaningful end points of biodiversity in life cycle assessment. *Environ. Sci. Technol.* **2011**, *45*, 70-79.
45. Hering, D.; Johnson, R.K.; Buffagni, A. Linking organism groups - major results and conclusions from the STAR project. *Hydrobiologia* **2006**, *556*, 109-113.
46. Hilty, J.; Merenlender, A. Faunal indicator taxa selection for monitoring ecosystem health. *Biol. Cons.* **2000**, *92*, 185-197.
47. Palmer, M.A.; Lettenmaier, D.P.; Poff, N.L.; Postel, S.L.; Richter, B.; Warner, R. Climate change and river ecosystems: protection and adaptation options. *Environmental Management* **2009**, *44*, 1053-1068.
48. *Watershed management: guiding principles for integrated management of water in Switzerland*; Swiss Federal Office for the environment: Bern, Switzerland, 2011; p 20.
49. Boulton, A.J.; Hancock, P.J. Rivers as groundwater-dependent ecosystems: a review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* **2006**, *54*, 133-144.
50. van Vliet, M.T.H.; Franssen, W.H.P.; Yearsley, J.R.; Ludwig, F.; Haddeland, I.; Lettenmaier, D.P.; Kabat, P. Global river discharge and water temperature under climate change. *Global Environmental Change* **2013**, *in press*.
51. Webb, B.W.; Hannah, D.M.; Moore, R.D.; Brown, L.E.; Nobilis, F. Recent advances in stream and river temperature research. *Hydrological Processes* **2008**, *22*, 902-918.
52. Caissie, D. The thermal regime of rivers: a review. *Freshwater Biology* **2006**, *51*, 1389-1406.
53. Gu, R.; Montgomery, S.; Al Austin, T. Quantifying the effects of stream discharge on summer river temperature. *Hydrological Sciences* **1998**, *43* (6), 885-904.
54. Zorn, T.G.; Seelbach, P.W.; Rutherford, P.W.; Wills, T.C.; Cheng, S.-T.; Wiley, M.J. *A regional-scale habitat suitability model to assess the effects of flow reduction on fish assemblages in Michigan streams*; Fisheries Division, Department of Natural Resources, State of Michigan, 2008; p 45.
55. Coutant, C.C. Compilation of temperature preference data. *Journal of the Fisheries Research Board of Canada* **1977**, *34* (5), 739-745.

56. Verones, F.; Hanafiah, M.M.; Pfister, S.; Huijbregts, M. A. J.; Pelletier, G.J.; Koehler, A. Characterization factors for thermal pollution in freshwater aquatic environments. *Environ. Sci. Technol.* **2010**, *44*, 9364-9369.
57. Markarian, R.K. A study of the relationship between aquatic insect growth and water temperature in a small stream. *Hydrobiologia* **1980**, *75* (81-95).
58. Floury, M.; Usseglio-Polatera, P.; Ferreol, M.; Delattre, C.; Souchon, Y. Global climate change in large European rivers: long-term effects on macroinvertebrate communities and potential local confounding factors. *Global Change Biology* **2013**, *19*, 1085-1099.
59. Burgmer, T.; Hillebrand, H.; Pfenninger, M. Effects of climate-driven temperature changes on the diversity of freshwater macroinvertebrates. *Oecologia* **2007**, *151*, 93-103.
60. Eaton, J.G.; Scheller, R.M. Effects of climate warming on fish thermal habitat in streams of the United States. *Limnol. Oceanogr.* **1996**, *41* (5), 1109-1115.
61. Notter, B. *Lebensraum der Bachforelle um 2050*; Swiss Federal Office for the Environment: Bern, Switzerland, 2009; pp 39-44.
62. Bartholow, J. *Stream Segment Temperature Model (SSTEMP) Version 2.0, Revised August 2002*; USGS: Fort Collins, USA, 2002.
63. Boyd, M.; Kasper, B. *Analytical methods for dynamic open channel heat and mass transfer: methodology for heat source model Version 7.0*; Oregon Department of Environmental Quality: Oregon, USA, 2003.
64. Bartholow, J.: *SNTEMP*, USGS, http://www.fort.usgs.gov/Products/Software/SNTEMP/, (2012).
65. Pelletier, G.J.; Chapra, S.C.; Tao, H. QUAL2Kw - a framework for modeling water quality in streams and rivers using a genetic algorithm for calibration. *Environmental Modelling and Software* **2006**, *21*, 419-425.
66. Wells, S.A. *Theoretical Basis for the CE-QUAL-W2 river basin model*; Dept. of Civil Eng., Portland St. Univ.: Portland, USA, 1997.
67. Haag, I.; Luce, A. The integrated water balance and water temperature model LARSIM-WT. *Hydrological Processes* **2008**, *22* (7), 1046-1056.
68. Caissie, D.; Satish, M.G.; El-Jabi, N. Predicting river water temperatures using the equilibrium temperature concept with application on Miramachi River catchments (New Brunswick, Canada). *Hydrological Processes* **2005**, *19*, 2137-2159.
69. Yearsley, John. A grid-based approach for simulating stream temperature. *Water Resources Research* **2012**, *48* (3), W03506.
70. Dübendorfer, C.; Moser, D.; Kempter, T.; Egloff, L.; Müller, V.; Wanner, P.; Kirchhofer, A.; Baumann, P. *Expertenbericht zu einem Modul Temperatur im Rahmen des Modul-Stufen-Konzepts*; BAFU: Bern, Switzerland, 2011; p 89.
71. Huijbregts, M. A. J.; Hellweg, S.; Hertwich, E. Do we need a paradigm shift in Life Cycle Impact Assessment? *Environ. Sci. Technol.* **2011**, *45*, 3833-3834.
72. *Water Framework Directive*. No. 2000/60/EC. European Parliament: Luxemburg, 2000.
73. Szerencsits, E.; Schüpbach, B.; Conradin, H.; Grünig, A.; Walter, T. *Agrarlandschaftstypen der Schweiz*; Agroscope ART: Zürich, Switzerland, 2009.
74. Gonseth, Y.; Wohlgemuth, T.; Buttler, A. *Die biogeographischen Regionen der Schweiz. Erläuterungen und Einteilungsstandard.*: Bern, Switzerland, 2001; p 48.
75. Vogt, J.; Soille, P.; de Jager, A.; Rimaviciute, E.; Mehl, W.; Foisneau, S.; Bodis, K.; Dusart, J.; Paracchini, M.L.; Haastrup, P.; Bamps, C. *A pan-European river and catchment database*; European Communities: Luxembourg, 2007.
76. Lubini, V.; Knispel, S.; Sartori, M.; Vincentini, H.; Wagner, A. *Rote Listen Eintagsfliegen, Steinfliegen, Köcherfliegen. Gefährdete Arten der Schweiz, Stand 2010.*: Bern and Neuenburg, Switzerland, 2012; p 111.

77. Dengler, J. Which function describes the species-area relationship best? A review and empirical evaluation. *J. Biogeogr.* **2009**, *36*, 728-744.
78. Xenopoulos, M.A.; Lodge, D.M. Going with the flow: using species-discharge relationships to forecast losses in fish biodiversity. *Ecology* **2006**, *87* (8), 1907-1914.
79. Tjorve, E. Shapes and functions of species-area curves: a reveiew of possible models. *Journal of Biogeography* **2003**, *30*, 827-835.
80. McGarvey, D.J. Quantifying ichtyofaunal zonation and species richness along a 2800-km reach of the Rio Chana and Rio Grande (USA). *Ecology of Freshwater Fish* **2011**, *20*, 231-242.
81. Matthews, W.J. *Patterns in Freshwater Fish Ecology*; Thomson: 1998.
82. Huet, M. Aperçu des relations entre la pente et les populations piscicoles des eaux courantes. *Schweizerische Zeitschrift für Hydrologie* **1949**, *11*, 333-351.
83. R-project. http://www.r-project.org/.
84. Verones, F.; Saner, D.; Pfister, S.; Baisero, D.; Rondinini, C.; Hellweg, S. Effects of consumptive water use on biodiversity in wetlands of international importance. *Submitted* **2013**.
85. IUCN Red List. http://www.iucnredlist.org (2012).
86. IUCN. *IUCN Red List categories and criteria, version 3.1*: Gland, Switzerland, 2000.
87. Purvis, A.; Gittleman, J.L.; Brooks, T., Eds. *Phylogeny and Conservation*. Cambridge University Press: Cambridge, UK, 2005.
88. Swiss Center for Biological Records. http://www.cscf.ch/ (18.05.2011).
89. Global Biodiversity Information Facility. http://data.gbif.org (27.01.2012).
90. Pfaundler, M.; Massimiliano, Z. *Die mittleren Abflüsse über die ganze Schweiz - Ein optimierter Datensatz im 500x500 m Raster*; Swiss Federal Office for the Environment: Bern, Switzerland, 2006.
91. Global Runoff Database. http://www.bafg.de/GRDC/EN/Home/homepage__node.html (05.05.2011).
92. European Water Archive. http://www.bafg.de/GRDC/EN/04_spcldtbss/42_EWA/ewa.html (05.05.2011).
93. Environment, Swiss Federal Office for the. Einzugsgebietsgliederung der Schweiz. http://www.bafu.admin.ch/hydrologie/01835/11452/index.html?lang=de.
94. Joint Research Center CCM Project. http://ccm.jrc.ec.europa.eu/php/index.php?action=view&id=23 (24.01.2012).
95. Weissmann, H.Z.; Könitzer, C.; Bertiller, A. *Ecomorphologie des cours d'eau suisses. Etat du lit, des berges et des rives. Résultats des relevés écomorphologiques (avril 2009)*; Swiss Federal Office for the Environment: Bern, Switzerland, 2009.
96. *Dataset: SwissAlti3D*; Swisstopo: Bern, Switzerland, 2012, http://www.swisstopo.admin.ch/internet/swisstopo/en/home/products/height/swissALTI3D.html.
97. *Dataset: Elevation map of Europe*; European Environmental Agency: Copenhagen, Denmark, 2004, http://www.eea.europa.eu/data-and-maps/data/digital-elevation-model-of-europe.
98. Drakare, Stina; Lennon, Jack J.; Hillebrand, Helmut. The imprint of the geographical, evolutionary and ecological context on species–area relationships. *Ecol. Lett.* **2006**, *9* (2), 215-227.
99. Oberdorff, T.; Hugueny, B.; Guégan, J.-F. Is there an influence of historical events on contemporary fish species richness in rivers? Comparisons between Western Europe and North America. *Journal of Biogeography* **1997**, *24*, 461-457.
100. McGarvey, D.J.; Hughes, R.M. Longitudinal zonation of Pacific Northwest (USA) fish assemblages and the species-discharge relationship. *Copeia* **2008**, *2*, 311-321.
101. Matthews, W.J.; Marsh-Matthews, E. Effects of drought on fish across axes of space, time and ecological complexity. *Freshwater Biology* **2003**, *48*, 1232-1253.

102. Detenbeck, N.E.; DeVore, P.W.; Niemi, G.J.; Lima, A. Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. *Environmental Management* **1992**, *16* (1), 33-53.
103. Dewson, Z.S.; James, A.B.W.; Death, R.G. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. *J. N. Am. Benthol. Soc.* **2007**, *26* (3), 401-415.
104. Petts, G.E.; Amoros, C., Eds. *Fluvial Hydrosystems*. Chapman and Hall: London, UK, 1996.
105. Lake, P.S. Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology* **2003**, *48*, 1161-1172.
106. Gorski, S. *In-stream temperature measurements and modeling of the Broye catchment*. Master Thesis, ETHZ, Zürich, Switzerland, 2012.
107. Deas, M.L.; Lowney, C.L. *Water temperature modeling review*; California Water Modeling Forum: Central Valley, USA, 2000.
108. iWaQa project. http://www.snf.ch/nfp/61/E/projects/cluster-water-management/integrated-river_water-quality_management/Pages/default.aspx.
109. *Dataset: Arealstatistik der Schweiz*; Swiss Federal Office for Statistics: Neuchâtel, Switzerland, 2004.
110. *Dataset: GEOSTAT: Soil suitability map of Switzerland*; Swiss Federal Office for Statistics: Neuchâtel, Switzerland, 2012.
111. Meteoswiss. http://www.meteosuisse.admin.ch/web/en/services/data_portal/idaweb.html (2012).
112. Robra, J. P.; Mastrullo, J. *Evaluation des besoins en eau d'irrigation dans la Broye*; MandaTerre: Yverdon, Switzerland, 2011.
113. Bowler, D.; Hannah, D.M.; Orr, H.; Pullin, A.S. What are the effects of wooded riparian zones on stream temperatures and stream biota? *Environmental Evidence* **2012**, *1* (3), 1-9.
114. Hrachowitz, M.; Soulsby, C.; Imholt, C.; Malcolm, I.A.; Tetzlaff, D. Thermal regimes in a large upland salmon river: a simple model to identify the influence of landscape controls and climate change on maximum temperatures. *Hydrological Processes* **2010**, *24*, 3374-3391.
115. Welty, J.J.; Beechie, T.; Sullivan, K.; Hyink, D.M.; Bilby, R.E.; Andrus, C.; Pess, G. Riparian aquatic interaction simulator (RAIS): a model of riparian forest dynamics for the generation of large woody debris and shade. *Forest Ecology and Management* **2002**, *162*, 299-318.
116. Honti, M. Eawag, Zürich, Switzerland. Personal communication (18.02.2013).
117. ISO, Environmental Management - Life Cycle Assessment - Principles and Framework. In International Organisation for Standardisation: Geneva, Switzerland, 2006a.
118. *Dataset: Swiss federal hydrological database*; Swiss Federal Office for the Environment: Bern, Switzerland, 2010, http://www.hydrodaten.admin.ch/e/index.htm.
119. Calanca, P.; Semenov, M.A. Local-scale climate scenarios for impact studies and risk assessments: integration of early 21st century ENSEMBLES projections into the ELPIS database. *Theor Appl Climatol* **2012**; DOI 10.1007/s00704-012-0799-3.
120. *Dataset: Hydrologische Messdaten*; Amt für Abfall, Wasser, Energie und Luft: Zürich, Switzerland, 2010, http://www.zh.ch/internet/baudirektion/awel/de/wasserwirtschaft/messdaten.html.
121. *ENSEMBLES: Climate change and its impacts: summary of research and results from the ENSEMBLES project*; Met Office: Exeter, UK, 2009; p 160.
122. Hayashi, K. Practical recommendations for supporting agricultural decisions through life cycle assessment based on two alternative views of crop production: the example of organic conversion. *Int J LCA* **2013**, *18*, 331-339.
123. Bockstaller, C.; Guichard, L.; Keichinger, O.; Girardin, P.; Galan, M.-B.; Gaillard, G. Comparison of methods to assess the sustainability of agricultural systems. A review. *Agron. Sustain. Dev.* **2009**, *29*, 223–235.

124. Frischknecht, Rolf; Jungbluth, Niels; Althaus, Hans-Jörg; Doka, Gabor; Hellweg, Stefanie; Hischier, R.; Nemecek, Thomas; Margni, M.; Spielmann, M. *Implementation of life cycle assessment methods - ecoinvent data v1.1*; Swiss Centre for Life Cycle Inventories (ecoinvent): Dübendorf, Switzerland, 2004; p 116.
125. Hauschild, M.; Potting, J. *Spatial differentiation in Life Cycle Impact Assessment - The EDIP2003 methodology*; Technical University of Denmark and Danish Ministry of the Environment: Denmark, 2005; p 195.
126. Rosenbaum, R.K.; Bachmann, T.M.; Gold, L.S.; Huijbregts, M. A. J.; Jolliet, O.; Juraske, R.; Koehler, A.; Larsen, H.F.; MacLeod, M.; Margni, M.; McKone, T.E.; Payet, J.; Schuhmacher, M.; van de Meent, D.; Hauschild, M. Z. USEtox - the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. *Int J LCA* **2008**, *13*, 532-546.
127. Guinée, J. B.; Gorrée, M.; Heijungs, R.; Huppes, G.; Kleijn, R.; de Koning, A.; van Oers, L.; Wegener Sleeswijk, A.; Suh, S.; Udo de Haes, H.A.; de Bruijn, H.; van Duin, R.; Huijbregts, M.A.J.; Lindeijer, E.; Roorda, A.A.H.; Weidema, B.P. *Life cycle assessment - An operational guide to the ISO standards*; Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML): Den Haag and Leiden, Netherlands, 2001.
128. Tendall, D.M.; Hellweg, S.; Pfister, S.; Huijbregts, M. A. J.; Gaillard, G. Impacts of River Water Use on Aquatic Biodiversity in Life Cycle Assessment. *Environ. Sci. Technol.* **2013**, *submitted*.
129. Jeanneret, P.; Baumgartner, D.; Freiermuth, R.; Gaillard, G. *Méthode d'évaluation de l'impact des activités agricoles sur la biodiversité dans les bilans écologiques – SALCA-BD*; Agroscope FAL Reckenholz: Online at http://www.reckenholz.ch/doc/fr/forsch/control/bilanz/salca-bd.pdf, 2006; p 67.
130. Alig, M.; Gaillard, G.; Müller, G., In *LCM in agriculture: enhancing the self-responsibility of farmers*, Proceedings of 6th Int. Conf. on LCA in the Agri-Food Sector, Zürich, Switzerland, 2008; Nemecek, T.; Gaillard, G., Eds. Agroscope Reckenholz-Tänikon Research Station ART: pp 312-317.
131. Nemecek, T.; Huguenin-Elie, O.; Dubois, D.; Gaillard, G. *Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau*; Agroscope FAL Reckenholz: Zürich, Switzerland, 2005; p 155.
132. Roches, A.; Nemecek, T. *Unilever-ART project no. CH-2008-0779 on variability of bio-based materials: final report*; Agroscope Reckenholz-Tänikon ART: Zürich, Switzerland, 2009; p 108.
133. Pré Consultants: *SimaPro*, http://www.pre-sustainability.com/simapro-lca-software, (2013).
134. Flisch, R.; Sinaj, S.; Charles, R.; Richner, W. *GRUDAF 2009: principles for fertilisation in arable and fodder production (in German)*: Bern, Switzerland, 2009; p 100.
135. Kupper, T.; Bonjour, C.; Achermann, B.; Rihm, B.; Zaucker, F.; Menzi, H. *Ammonia emissions for Switerland 1990 to 2010 and previsions until 2020*; Bundesamt für Umwelt: Bern, Switzerland, 2013; p 110.
136. Spoerri, M. *Economic efficiency of irrigation strategies for arable crops under current and future climate scenarios*. ETHZ, Zürich, Switzerland, 2011.
137. Richner, W.; Oberholzer, H.-R.; Freiermuth, R.; Huguenin, O.; Walther, O. *Modell zur Beurteilung des Nitratauswaschungspotenzials in Ökobilanzen - SALCA-Nitrat*; Agroscope FAL Reckenholz: Zürich, Switzerland, 2006.
138. Prasuhn, V. *Erfassung der PO4-Austräge für die Ökobilanzierung: SALCA-Phosphor*; Agroscope FAL Reckenholz: Zürich, Switzerland, 2006.
139. Freiermuth, R. *Modell zur Berechnung der Schwermetallflüsse in der Landwirtschaftlichen Ökobilanz: SALCA-Schwermetall*; Agroscope FAL Reckenholz: Zürich, Switzerland, 2006.

140. Nemecek, T.; Heil, A.; Huguenin, O.; Meier, S.; Erzinger, S.; Blaser, S.; Dux, D.; Zimmermann, A. *Life cycle inventories of agricultural production systems - ecoinvent data v1.1*; Swiss Centre for Life Cycle Inventories: Dübendorf, Switzerland, 2004; p 289.
141. *ecoinvent Data - The Life Cycle Inventory Data V2.2*; Swiss Centre for Life Cycle Inventories: Dübendorf, Switzerland, 2010.
142. Hayer, F. Aquatic and terrestrial ecotoxicity as well as human toxicity characterisation factors for pesticide emissions to soil according to the methods USES-LCA and EDIP. Agroscope Reckenholz-Tänikon; Unpublished, 2010; p 2.
143. Tendall, D.M.; Binder, C.R. Nuclear energy in Europe: uranium flow modeling and fuel cycle scenario trade-offs from a sustainability perspective. *Environ. Sci. Technol.* **2011**, *45* (6), 2442-2449.
144. Lehmann, N. Modeling of farm adaptation to climate change scenarios in Switzerland: results for mixed farms. ETHZ; Zürich, Switzerland. Unpublished, 2013.
145. Huijbregts, M.A.J. Application of uncertainty and variability in LCA. *Int. J. LCA* **1998**, *3* (5), 273-280.
146. Vermeulen, Sonja J.; Challinor, Andrew J.; Thornton, Philip K.; Campbell, Bruce M.; Eriyagama, Nishadi; Vervoort, Joost M.; Kinyangi, James; Jarvis, Andy; Läderach, Peter; Ramirez-Villegas, Julian; Nicklin, Kathryn J.; Hawkins, Ed; Smith, Daniel R. Addressing uncertainty in adaptation planning for agriculture. *Proceedings of the National Academy of Sciences* **2013**, *110* (21), 8357-8362.
147. Huijbregts, M.A.J.; Thissen, U.; Jager, T.; van de Meent, D.; Ragas, A.M.J. Priority assessment of toxic substances in life cycle assessment. Part II: assessing parameter uncertainty and human variability in the calculation of toxicity potentials. *Chemosphere* **2000**, *41*, 575-588.
148. Lehmann, N. ETHZ, Zürich, Switzerland. Personal communication (20.02.2013).
149. Heller, M.C.; Keoleian, G.A., In *A novel nutrition-based functional equivalency metric for comparative life cycle assessment of food*, Proceedings of 8th Int. Conference on LCA in the Agri-Food Sector, St. Malo, 1-4 October, 2012; Corson, M.S.; Van der Werf, H., Eds.
150. Kernebeek, H.R.J.; Oosting, S.J.; de Boer, I.J.M., In *Comparing the environmental impact of human diets varying in amount of animal-source food - the impact of accounting for nutritional quality*, Proceedings of 8th Int. Conference on LCA in the Agri-Food Sector, St. Malo, 1-4 October, 2012; Corson, M.S.; Van der Werf, H., Eds.
151. Klein, T. PhD thesis, University of Bern, Bern, Switzerland, 2013.
152. Loiseau, E.; Roux, P.; Junqua, G.; Maurel, P.; Bellon-Maurel, V., In *Adaptation of the LCA framework to support land use planning policies*, Proceedings of SETAC Europe, Glasgow, 12 May - 16 May, 2013.
153. Gomez, M. *A groundwater flow model of the Broye catchment: effects of irrigation withdrawals on the environment*. Master thesis, ETHZ, Zürich, Switzerland, 2012.
154. *Dataset: Klimaeignungskarte für die Landwirtschaft in der Schweiz*; Agriculture, Swiss Federal Office of: Bern, Switzerland, 2012, http://www.blw.admin.ch/dienstleistungen/00334/00336/.
155. Ghilardi, T. *Renaturation de la Broye*. Master Thesis, EPFL, Lausanne, Switzerland, 2008.
156. Malcolm, I.A.; Soulsby, C.; Hannah, D.M.; Bacon, P.J.; Youngson, A.F.; Tetzlaff, D. The influence of riparian woodland on stream temperatures: implications for the performance of juvenile salmonids. *Hydrological Processes* **2008**, *22*, 968-979.
157. Johnson, S.L. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. *Can. J. Fish. Aquat. Sci.* **2004**, *61*, 913-923.
158. Hutchins, M.G.; Johnson, A.C.; Deflandre-Vlandas, A.; Comber, S.; Posen, P.; Boorman, D. Which offers more scope to suppress river phytoplankton blooms: reducing nutrient pollution or riparian shading? *Science of the Total Environment* **2010**, *408* (21), 5065-5077.

159. Ghermandi, A.; Vandenberghe, V.; Benedetti, L.; Bauwens, W.; Vanrolleghem, P.A. Model-based assessment of shading effect by riparian vegetation on river water quality. *Ecological Engineering* **2009**, *35*, 92-104.
160. Poole, G.C.; Berman, C.H. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. *Environmental Management* **2001**, *27* (6), 787-802.
161. Durance, I.; Ormerod, S.J. Trends in water quality and discharge confound long-term warming effects on river macroinvertebrates. *Freshwater Biology* **2009**, *54*, 388-405.
162. Kirchhofer, A. Morphological variability in the ecotone - an important factor for the conservation of fish species richness in Swiss rivers. *Hydrobiologia* **1995**, *303*, 103-110.
163. Watanabe, Michio; Adams, Richard M.; Wu, JunJie; Bolte, John P.; Cox, Matt M.; Johnson, Sherri L.; Liss, William J.; Boggess, William G.; Ebersole, Joseph L. Toward efficient riparian restoration: integrating economic, physical, and biological models. *Journal of Environmental Management* **2005**, *75* (2), 93-104.
164. Aquavision Engineering; Raymond Delarze; Hintermann & Weber; Mandaterre; Bureau Nicod+Perrin. *Diagnostic de la plaine de la Broye*; Etat de Vaud - SESA, Etat de Fribourg - SLCE, 2011; p 129.
165. Chinnayakanahalli, K.J.; Hawkins, C.P.; Tarboton, D.G.; Hill, R.A. Natural flow regime, temperature and the composition and richness of invertebrate assemblages in streams of the western United States. *Freshwater Biology* **2011**, *56*, 1248-1265.
166. Borsuk, M.E.; Reichert, P.; Peter, A.; Schager, E.; Burkhardt-Holm, P. Assessing the decline of brown trout (Salmo trutta) in Swiss rivers using a Bayesian probability network. *Ecological Modelling* **2005**, *192*, 224-244.
167. Iwasaki, Y.; Ryo, M.; Sui, P.; Yoshimura, C. Evaluating the relationship between basin-scale fish species richness and ecologically relevant flow characteristics in rivers worldwide. *Freshwater Biology* **2012**, *57*, 2173-2180.
168. Gaillard, G.; Nemecek, T., In *Swiss Agricultural Life Cycle Assessment (SALCA): An integrated environmental assessment concept for agriculture*, Proceedings of Int. Conf. "Integrated Assessment of Agriculture and Sustainable Development, Setting the Agenda for Science and Policy", Egmond aan Zee, The Netherlands, 2009; AgSAP Office, Wageningen University: pp 134-135.
169. Alder, S. *Modellierung des Gewässeranschlusses potenziell erosionsgefährdeter Flächen im schweizerischen Mittelland*. Master thesis, University of Bern, Bern, Switzerland, 2012.
170. Ellenberg, H. *Zeigerwerte der gefässpflanzen Mitteleuropas (in German)*; 2nd ed.; Goltze: Göttingen, Germany, 1979.
171. Landolt, E.; Bümler, B.; Erhardt, A.; Hegg, O.; Klötzli, F.; Lämmler, W.; Nobis, M.; Rudmann-Maurer, K.; Schweingruber, F.H.; Theurillat, J.-P.; Urmi, E.; Vust, M.; Wohlgemuth, T. *Flora Indicativa*; 2nd ed.; Haupt Verlag: Bern, Switzerland, 2010.
172. Klein, T. Greifensee agricultural adaptation scenarios with urban expansion. Agroscope Reckenholz-Tänikon ART; Zürich, Switzerland. Unpublished, 2012.
173. Griebler, C.; Stein, H.; Kellermann, C.; Berkhoff, S.; Brielmann, H.; Schmidt, S.; Selesi, D.; Steube, C.; Fuchs, A.; Hahn, H.-J. Ecological assessment of groundwater ecosystems - vision or illusion? *Ecological Engineering* **2010**, *36*, 1174-1190.
174. Harris, G.P.; Heathwaite, A.L. Why is achieving good ecological outcomes in rivers so difficult? *Freshwater Biology* **2011**; 10.1111/j.1365-2427.2011.02640.x.
175. Woodward, G.; Perkins, D.M.; Brown, L.E. Climate change and freshwater ecosystems: impacts across multiple levels of organization. *Phil. Trans. R. Soc. B* **2010**, *365*, 2093-2106.

176. Yang, Y.-C. E.; Cai, X.; Herricks, E.E. Identification of hydrologic indicators related to fish diversity and abundance: a data mining approach for fish community analysis. *Water Resources Research* **2008**, *44*.
177. Lancaster, J.; Downes, B.J. Linking the hydraulic world of individual organisms to ecological processes: putting ecology into ecohydraulics. *River Res. Applic.* **2009**, *26*, 385-403.
178. Heino, J.; Virkkala, R.; Toivonen, H. Climate change and freshwater biodiversity: detected patterns, future trends and adaptations in northern regions. *Biol. Rev.* **2009**, *84*, 39-54.
179. Schmidt, J.H. System delimitation in agricultural consequential LCA. *Int J LCA* **2008**, *13*, 350-364.
180. Sonnevelt, M.; Aepli, M. ETHZ, Zürich, Switzerland. Personal communication (17.05.2013).
181. *Schweizerische Aussenhandelsstatistik 5.2: Ein- und Ausfuhr landwirtschaftlicher Erzeugnisse im Jahr 2011*; Schweizerische Bauernverband: Switzerland, 2011.
182. Alexandratos, N.; Bruinsma, J. *World agriculture towards 2030/2050*; FAO: Rome, Italy, 2012.
183. FAO. FAOStat. http://faostat.fao.org/ (23.05.2013).
184. FAO. Agricultural outlook. http://www.oecd.org/site/oecd-faoagriculturaloutlook/database-oecd-faoagriculturaloutlook.htm (23.05.2013).
185. Park, S. E.; Howden, S. M.; Crimp, S. J.; Gaydon, D. S.; Attwood, S. J.; Kokic, P. N. More than Eco-efficiency is Required to Improve Food Security. *Crop Science* **2010**, *50* (2), S132-S141.
186. Wood, R.; Hertwich, E.G. Economic modelling and indicators in life cycle sustainability assessment. *Int J LCA* **2013**; DOI 10.1007/s11367-012-0463-2.
187. Kissinger, G.; Brasser, A.; Gross, L. *Reducing risk: landscape approaches to sustainable sourcing*; Landscapes for People, Food and Nature Initiative: Washington DC, USA, 2013; p 131.
188. Yang, Yi-Chen. *Modeling watershed management with an ecological objective - a multi-agent system based approach*. PhD Thesis, University of Illinois at Urbana-Champaign, Urbana, USA, 2010.
189. Webb, J.; Williams, A.G.; Hope, E.; Evans, D.; Moorhouse, E. Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK? *Int J LCA* **2013**; DOI 10.1007/s11367-013-0576-2.
190. Kleijn, D.; Kohler, F.; Báldi, A.; Batáry, P.; Concepción, E.D.; Clough, Y.; Díaz, M.; Gabriel, D.; Holzschuh, A.; Knop, E.; Kovács, A.; Marshall, E.J.P.; Tscharntke, T.; Verhulst, J. On the relationship between farmland biodiversity and land-use intensity in Europe. *Proc. R. Soc. B* **2009**, *276*, 903-909.
191. Reidsma, P.; Tekelenburg, T.; van den Berg, M.; Alkemade, R. Impacts of land-use change on biodiversity: an assessment of agricultural biodiversity in the European Union. *Agriculture, Ecosystems & Environment* **2006**, *114*, 86-102.
192. Tilman, D.; Balzer, C.; Hill, J.; Befort, B.L. Global food demand and the sustainable intensification of agriculture. *PNAS* **2011**, *108* (50), 20260-20264.
193. Hoekstra, A.Y.; Chapagain, A.K.; Aldaya, M.M.; Mekonnen, M.M. *Water Footprint Manual*: Enschede, 2009.
194. Schager, E.; Peter, A. *Methoden zur Untersuchung und Beurteilung der Fliessgewässer: Fische, Stufe F*; Swiss Federal Office for the Environment: Bern, Switzerland, 2004; p 63.
195. FishBase. http://www.fishbase.org/ (2012).
196. Honti, M. Water temperature model. Eawag; Dübendorf, Switzerland. Unpublished, 2011-2013.
197. Yearsley, J.R. A semi-lagrangian water temperature model for advection-dominated river systems. *Water Resources Research* **2009**, *45* (12), W12405.
198. Honti, M.; Istvanovics, V.; Kovacs, A.S. Balancing between retention and flushing in river networks - optimizing nutrient management to improve trophic state. *Science of the Total Environment* **2010**, *408*, 4712-4721.

199. Kovacs, A.; Honti, M.; Zessner, M.; Eder, A.; Clement, A.; Blöschl, G. Identification of phosphorus emission hotspots in agricultural catchments. *Science of the Total Environment* **2012**, *433*, 74-88.
200. Leopold, L.B.; Wolman, M.G.; Miller, J.P. *Fluvial processes in geomorphology*; W.H. Freeman and Co.: San Francisco, USA, 1964.
201. Liu, Y.B.; De Smedt, F. *WetSpa Extension: A GIS-based hydrologic model for flood prediction and watershed management*; Vrije Universiteit Brussel: Brussels, Belgium, 2004.
202. Bera, M.; Borah, D.K. Watershed-scale hydrologic and non-point source pollution models: review of mathematical bases. *Transactions of the ASAE* **2003**, *46*, 1553-1566.
203. Fread, D.L. *Flow routing*; McGraw-Hill: New York, USA, 1993.
204. Edinger, J.E.; Duttweiler, D.W.; Geyer, J.C. The response of water temperatures to meteorological conditions. *Water Resources Research* **1968**, *4* (5), 1137-1143.
205. Zheng, D.; Hunt, R. Jr.; Running, S.W. A daily soil temperature model based on air temperature and precipitation for continental applications. *Climate Research* **1993**, *2*, 183-191.
206. Sinokrot, B.A.; Stefan, H.G. Stream temperature dynamics: measurements and modeling. *Water Resources Research* **1993**, *29* (7), 2299-2312.
207. Molnar, P. Watershed modeling: calibration and validation of hydrological models. ETH Zürich, Switzerland; Zürich, Switzerland. Unpublished, 2012.
208. Ridoutt, B. G.; Pfister, S. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. *Global Environmental Change-Human and Policy Dimensions* **2009**, *20* (1), 113-120.
209. Kennedy, J.; Eberhart, R. Particle swarm optimization. *Neural Networks* **1995**, *4*, 1942-1948.
210. McKinnon, K.I.M. Convergence of the Nelder-Mead simplex method to a non-stationary point. *J. on Optimization* **1998**, *9* (1), 148-158.
211. Beven, K.J. *Rainfall-Runoff modelling: the primer*; John Wiley and Sons Ltd: Chichester, UK, 2001.
212. Metropolis, N.; Rosenbluth, A.W.; Rosenbluth, M.N.; Teller, A.H.; Teller, E. Equations of state calculations by fast computing machines. *J. Chem. Phys.* **1953**, *21*, 1087-1091.
213. Hastings, W.K. Monte Carlo sampling methods using Markov chains and their applications. *Biometrika* **1970**, *57*, 97-109.
214. Kovacs, A.S.; Honti, M.; Clement, A. Design of best management practice applications for diffuse phosphorus pollution using interactive GIS. *Water Science and Technology* **2008**, *57* (11), 1727-1733.
215. Meier, W.; Bonjour, C.; Wuest, A.; Reichert, P. Modeling the effect of water diversion on the temperature of mountain streams. *Journal of Environmental Engineering* **2003**, 755-764.
216. Lehmann, N. Working paper: AGWAM livestock model. ETHZ; Zürich, Switzerland. Unpublished, 2013.
217. Stöckle, C.; Donatelli, M.; Nelson, R. CropSyst, a cropping systems simulation model. *European Journal of Agronomy* **2003**, *18* (3-4), 289-307.
218. Semenov, M.A.: *LARS-WG*, Rothampstead, UK, http://www.rothamsted.ac.uk/mas-models/larswg.html, (1991).

ANNEX 1: LIFE CYCLE IMPACT ASSESSMENT METHOD FOR RIVER WATER CONSUMPTION

A1.1. Detailed watershed-level species-discharge relationship results

In the case of Switzerland, the Swiss SDR resulted in the use of 769 watersheds, with annual average discharges ranging from 0.24 to 576 m^3/s. The watersheds are partly nested, with smaller watersheds often part of larger watersheds. For Europe, watersheds consist of basins with outlet to the sea and are therefore entirely non-nested; the 399 European watersheds have average annual discharges ranging from 0.1 to 6434 m^3/s.

When using species-discharge relationships specific to eco-regions, goodness of fit (measured by Pearson's R^2) increases from 0.69 for whole of Switzerland to 0.9 for the lowland region (where most agricultural irrigation activity is located). The R^2 for the alpine region is on the contrary reduced to 0.58, suggesting that influence of other environmental parameters on species richness may dominate over discharge. It can be noted that for small discharges, there is a higher variability among the observed species richness, which may be interpreted as causing a higher uncertainty in the prediction of an effect of changes in discharge for the case of smaller rivers. This higher variability for small watersheds is an effect commonly observed in species-area relationships [77].

R^2 is further improved to 0.93 by using the cumulative Weibull function, capable of reproducing the asymptotical behavior of the species-discharge relationship in Switzerland. Although the improvement is marginal, this function visibly fits the data better. Accounting for over-fitting, the cumulative Weibull function is significantly better ($F=54.44$, $p<0.001$) using the F-test (which assesses goodness of fit while accounting for increase in degree of freedom), and the AIC is also lower (1075 for Weibull, resp. 1119 for power function). As a further statistical validation, having calibrated the fit on a random subset containing 70% of the data, goodness of fit ($R^2 = 0.92$) remained constant when applying this parameterization to the remaining 30% of the data (data split validation).

EPT species richness in Switzerland shows a relationship to discharge similar to that of fish, although maximum EPT species richness is higher than maximum fish species richness in Switzerland (317 resp. 45 species observed). Therefore according to the species-discharge relationship principle, it may be expected that EPTs will be affected similarly to fish in the event of reductions in discharge due to increased water consumption. In other words, fish may be an appropriate indicator to represent impacts of water consumption on both fish and macro-invertebrates, if using relative species loss (as in the pre-existing method). However, in the advocated method, absolute species loss is used. Therefore it is

desirable to include as many taxa groups as possible, and if only one taxa group (such as fish) is considered, it must be kept in mind that this is a lower estimate, since further species from other taxa groups may also be affected.

The species-discharge relationship at the European continental scale only achieves an R^2 of 0.35, which is further reduced to 0.23 when the outlying Danube basin is not considered. This is significantly lower than the goodness of fit achieved for the Swiss species-discharge relationship. This supports the principle that variability is reduced when using spatial subsets of data (eg. Eco-regions). However it may also be due to inhomogeneous data from different sources with varying coverage and quality in the case of Europe. Similarly to Switzerland, when developing species-discharge relationships per eco-region, certain regions benefit from an improvement in goodness of fit (such as the Iberic and Central Plains regions), whereas others show no significant relationship (such as the United Kingdom). Differing ranges of discharge, species present, data quality and coverage may cause a part of these differences, however this may also indicate, as in Switzerland, that species richness is more clearly influenced by discharge in certain regions than in others: for the latter, the prediction of changes in species richness due to changes in discharge is therefore accompanied by higher uncertainty.

A1.2. Threat status conversion from IUCN to linear scale

Table A 1.1 shows the conversion of the qualitative IUCN threat status levels to a linear scale from 1 to 5.

Table A1.1: *Conversion of IUCN threat status of species [86] to the scale used for threat status weighting in this thesis.*

IUCN threat status	Threat status weighting factor
Least concern (LC)	1
Near threatened (NT)	2
Vulnerable (VU)	3
Endangered (EN)	4
Critically endangered (CR)	5

A1.3. Zones used to develop the Swiss zone-level species-discharge relationship

Table A1.2 provides the discharge, area, attributed zone class and species richness of the zones used to develop the Swiss zone-level species-discharge relationship for fish.

Table A1.2: *Yearly average discharge, area, attributed zone class according to Huet et al. (1949) [82] and fish species richness (number of species observed) for the zones used to develop the zone-level species-discharge relationship of fish for Switzerland.*

Zone ID	Discharge (m^3/s)	Area (m^2)	Attributed zone	Fish species richness
Laire	4.12E-01	4.35E+07	Trout	9
Lion	1.07E+00	1.15E+08	Trout	9
Versoix	6.08E-02	8.45E+06	Barbel	3
Promenthouse	4.28E+00	8.64E+07	Trout	4
Aubonne	2.72E+00	8.26E+07	Trout	17
Venoge	2.91E+00	3.09E+07	Grayling	15
Boiron	5.73E-01	2.41E+07	Trout	3
Orbe	9.40E-01	5.51E+06	Grayling	8
Orbe	9.40E-01	2.77E+07	Trout	11
Veyron	1.26E-01	7.98E+06	Grayling	3
Venoge	4.21E+00	7.60E+07	Trout	4
Morges	6.69E-01	3.85E+07	Trout	3
Senoge	3.14E-01	2.34E+07	Trout	5
Molombe	1.20E-01	1.01E+07	Trout	1
Sorge	2.21E-01	7.10E+06	Grayling	2
Mèbre	3.39E-01	2.20E+07	Trout	2
Talent	9.71E-01	6.15E+07	Trout	12
Nozon	5.90E-01	6.35E+07	Trout	8
Orbe	1.23E+01	9.40E+07	Trout	10
Tièhle	1.16E+01	4.47E+07	Barbel	22
Mujon	1.50E-01	1.28E+07	Trout	1
Brine	2.08E-01	1.40E+07	Trout	11
Arnon	2.52E+00	5.19E+07	Trout	11
Areuse	5.88E+00	1.98E+07	Grayling	12
Merdasson	3.26E-01	1.45E+07	Trout	1
Seyon	9.07E-01	5.20E+07	Trout	3
Bied du Locle	2.70E-01	2.50E+07	Trout	1
Rhône	1.84E+02	2.23E+08	Grayling	22
Thielle	2.71E+00	8.24E+07	Trout	24
Gryonne	9.28E-01	3.44E+07	Trout	23
Grande Eau	3.89E+00	1.44E+08	Trout	25
Vièze	3.74E+00	1.42E+08	Trout	16
Trient	6.11E+00	1.45E+08	Trout	25
Borgne	1.22E+01	3.85E+08	Trout	2
Navisence	7.98E+00	2.17E+08	Trout	2
Turtmänna	2.56E+00	9.34E+07	Trout	2

Rhine	1.05E+03	2.36E+07	Bream	42
Zihl	4.07E+01	6.59E+06	Bream	24
Gryonne	2.37E-01	1.74E+07	Barbel	23
Grande Eau	2.12E-01	4.36E+07	Barbel	25
Vièze 1	2.03E+00	1.45E+08	Trout	16
Aare	1.32E+02	7.79E+05	Bream	25
Aare	1.27E+02	3.67E+07	Grayling	29
Aare	1.32E+02	1.50E+08	Trout	25
Liène	3.56E+00	7.59E+07	Trout	3
Töss	9.96E+00	2.84E+07	Grayling	24
Aare	5.38E+02	1.15E+07	Barbel	32
Limmat	1.03E+02	4.16E+07	Grayling	30
Aare 2	3.26E+02	7.72E+07	Grayling	32
Suhre	4.18E+00	2.69E+07	Grayling	21
Rhine	1.56E+02	4.45E+09	Trout	20
Reuss	4.47E+01	8.31E+08	Trout	14
Giessbach	1.47E+00	2.62E+07	Trout	3

A1.4. Rarity and vulnerability weighting of species

Table A1.3 shows the rarity and threat status of the species considered in the case study (species occurring in the affected zones).

Table A1.3: *Rarity factors, threat status and zone attribution of the species considered in the case study (the zones are depicted in Figure 5).*

Species	Rarity factor	Threat status	Attributed zone
Abramis brama	2.62E-03	1	Bream
Alburnus alburnus	2.73E-03	1	Bream
Alburnoides bipunctatus	1.34E-04	1	Grayling
Alosa alosa	6.26E-03	1	Bream
Alosa fallax	5.22E-03	1	Bream
Ambloplites rupestris	1.41E-04	1	Barbel
Ameiurus melas	8.94E-04	1	Bream
Ameiurus nebulosus	1.47E-03	1	Bream
Anguilla anguilla	1.23E-03	5	Bream
Aspius aspius	1.43E-02	1	Bream
Barbatula barbatula	4.98E-05	1	Grayling

Barbus barbus	4.54E-04	1	Barbel
Blicca bjoerkna	3.75E-03	1	Bream
Carassius auratus	6.17E-05	1	Barbel
Carassius carassius	2.53E-04	1	Barbel
Carassius gibelio	4.48E-04	1	Grayling
Pseudochondrostoma nasus	7.61E-04	1	Barbel
Cobitis taenia	3.78E-03	1	Bream
Coregonus lavaretus	1.08E-02	3	Bream
Cottus gobio	1.64E-04	1	Trout
Ctenopharyngodon idella	1.16E-03	1	Bream
Cyprinus carpio	4.11E-04	3	Bream
Esox lucius	7.26E-05	1	Barbel
Gasterosteus aculeatus	8.88E-04	1	Bream
Gobio gobio	2.80E-04	1	Barbel
Gymnocephalus cernua	2.89E-03	1	Bream
Hypophthalmichthys molitrix	7.22E-05	2	Grayling
Lampetra fluviatilis	8.44E-05	1	Grayling
Lampetra planeri	1.56E-04	1	Trout
Lepomis gibbosus	1.14E-03	1	Bream
Leucaspius delineatus	5.89E-03	1	Bream
Leuciscus idus	4.18E-03	1	Bream
Leuciscus leuciscus	5.95E-05	1	Grayling
Lota lota	1.23E-04	1	Barbel
Micropterus salmoides	4.80E-04	1	Bream
Misgurnus fossilis	2.33E-02	1	Bream
Oncorhynchus mykiss	5.53E-04	1	Bream
Osmerus eperlanus	6.03E-03	1	Bream
Perca fluviatilis	1.79E-04	1	Barbel
Petromyzon marinus	1.87E-03	1	Bream
Phoxinus phoxinus	4.45E-05	1	Grayling
Platichthys flesus	2.81E-03	1	Bream
Poecilia reticulata	4.56E-05	1	Grayling
Pseudorasbora parva	2.64E-04	1	Barbel
Pungitius pungitius	1.19E-03	1	Bream
Rhodeus amarus	2.89E-02	1	Bream

Romanogobio albipinnatus	5.17E-03	1	Barbel
Rutilus rutilus	2.35E-04	1	Barbel
Salvelinus alpinus	1.11E-04	1	Trout
Salmo salar	4.34E-05	1	Trout
Salvelinus fontinalis	1.01E-04	1	Trout
Salvelinus namaycush	6.71E-05	1	Trout
Salmo trutta fario	8.23E-05	1	Trout
Salmo trutta lacustris	8.35E-05	1	Trout
Salmo trutta trutta	8.35E-05	1	Trout
Sander lucioperca	3.32E-03	1	Bream
Scardinius erythrophthalmus	1.73E-03	1	Bream
Silurus glanis	5.46E-04	1	Barbel
Squalius cephalus	2.61E-04	1	Barbel
Telestes souffia	1.42E-03	1	Barbel
Thymallus thymallus	6.93E-05	1	Grayling
Tinca tinca	1.46E-03	1	Bream
Vimba vimba	1.13E-02	1	Bream

Species without a threat status defined by the IUCN were given a factor 1: this is not a conservative estimate, however due to the impossibility of justifying any other factor, a neutral default factor that would not over-weight the results was preferred. The attribution of species to zones was based on several literature sources, in order of preference: explicit literature [104, 194], maps of occurrences [88, 89], or deduced from descriptions of habitat [195]; species occurring in several zones were attributed to the zone were they dominantly occur; if they occurred in all zones equally, species were attributed to the largest zone they occur in (causing a "conservative" estimate).

A1.5. Conversion between GSE*y and PDF*m^3*y

The impact units used by Hanafiah et al. are PDF*m^3*y, which is the fraction lost of total species in a local ecosystem (e.g. a river basin), weighted by the total volume of the local ecosystem. This method cannot relate the loss to an equivalent of global species extinction, and it does not weight species by their rarity or vulnerability. It also gives more weight to species-poor ecosystems (they obtain a high PDF even if only a small number of species is actually lost). In summary, it shows the relative species loss (as a fraction) and the absolute amount of habitat affected. Re-writing the units with a common denomination yields (Eq. A1.1):

$$PDF \cdot m^3 \cdot y = \frac{LSL}{LSR} LV \cdot t \qquad \text{(A1.1)}$$

With LSL = local species loss (in the entire river), LSR = local species richness (in the entire river) and LV = local volume (entire river), t = duration of impact.

GSE*y is the local species lost weighted by their areal rarity (the fraction of their habitat which is affected, or area of their habitat which is affected divided by their total habitat area), and their threat status. A global extinction occurs if all of their habitat is affected. This approach gives more weight to species-rich ecosystems (where more species can disappear). It also reflects location of withdrawal within the basin by aggregating the impacts for all downstream longitudinal zones. Assuming however also (for comparison's sake) that only one zone is concerned in this example: GSE*y re-written in the common denomination yields Eq. A1.2:

$$GSE \cdot y = LSL \frac{\sum\left(\frac{LSH}{TSH} * TS\right)}{LSR} \cdot t \qquad \text{(A1.2)}$$

With LSH = local species habitat area affected (per species) and TSH = total species habitat area (per species), the ratio of which is summed for all species in the local ecosystem affected.

A conversion factor X from GSE*y to PDF*m^3*y would thus be (Eq. A1.3 to A1.5):

$$\frac{LSL}{LSR} LV \cdot t = X \cdot LSL \frac{\sum\left(\frac{LSH}{TSH} * TS\right)}{LSR} \cdot t \qquad \text{(A1.3)}$$

$$LV = X \cdot \sum\left(\frac{LSH}{TSH} * TS\right) \qquad \text{(A1.4)}$$

$$X = \frac{LV}{\sum\left(\frac{LSH}{TSH} * TS\right)} \qquad \text{(A1.5)}$$

This conversion factor for the above example is the total volume of the river divided by the sum of "area of habitat affected, divided by total area of habitat, times the threat status" for each species in the river. (Note that this would be more complex for an ecosystem composed of several zones).

According to the results of the case study, the impacts in PDF*m^3*y (measured using the method and species-discharge relationship from Hanafiah et al.) are approximately a factor 2.69E+09 times higher than the impacts in GSE*y (this is using the full method proposed, with zone-level species-discharge relationship, rarity and threat status weighting, and aggregation of impacts in downstream zones: in Table 4, comparing row 1 in GSE*y with row 2 in PDF*m^3*y). However this is just one case study: any potential consistency in this ratio would have to be verified using many other case studies.

The LCIA method ReCiPe [26] uses species density in order to aggregate impacts in different ecosystems (PDF*m^3*y for aquatic and PDF*m^2*y for terrestrial). Species density is the total number of species in an ecosystem divided by the total volume or area of the ecosystem. This can also be seen as an estimate of the conversion between GSE*y and PDF*m^3*y: assuming a loss of all freshwater species (total number species = 100'000 according to ReCiPe) in all freshwater ecosystems (total volume = 2.63E+14 m^3 according to ReCiPe), PDF = 1, the average share of habitat affected = 1, average normalized threat status of species assumed to be 1 (if all are driven to extinction). The conversion factor X would then be (Eq. A1.6):

$$X = \frac{PDF \cdot m^3 \cdot y}{GSE \cdot y} = \frac{m^3}{SL} = \frac{2.63 \cdot 10^{14}}{100'000} = 2.63 \cdot 10^9 \quad \text{(A1.6)}$$

Where SL is the number of species lost. This is very similar to the conversion factor calculated above for the case study (2.69E+09). This conversion factor can also be interpreted as an equivalent of ecosystem amount that must be destroyed to cause the equivalent of a global extinction of one species.However, this is again just an extreme example: this ratio would have to be verified using many other case studies.

ANNEX 2: WATER TEMPERATURE MODEL

A2.1. Model description

The work of Mark Honti as developder of the iWaQa temperature model and as source of the model description information is acknowledged here [196]. The work of Gorski (2012) [106] (co-supervised by the author of this thesis) is acknowledged as the basis from which this model description is adapted.

The iWaQa temperature model is a one-dimensional semi-distributed deterministic model. In other words, it treats the channel flow as a series of individual water packages (hence one-dimensional) subjected to the driving parameters during the time they are present in the channel (residence time). Assuming plug flow, the conservation of thermal energy is solved for each water package using a semi-Lagrangian numerical scheme [197]. The watershed is divided into sub-catchments of homogeneous characteristics, and water temperature is reported at the outflow of each (hence semi-distributed). This represents an adequate compromise between precision and spatial coverage.

In order to model the water temperature within a watershed, five steps are necessary: (1) Mapping of flow routing and spatially distributed average flow and travel time in a GIS. This requires the following spatially explicit inputs: topography (elevation and slope), half-year precipitation sum, half-year average wind speed, half-year average air temperature, land use, soil class, and sub-catchment delineation. (2) Based on the flow routing, average flow and travel time, the prior residence time to depth (or tau/d) ratio is calculated for each sub-catchment outlet. This represents the characteristic of each water package required for calculation of its temperature (time during which it is subjected to the external drivers, and water depth essential for calculating heat exchange). Using in addition the land use map, this step also estimates the prior channel shading coefficient for each station, based on the proportion of channel that crosses forested land. (3) The other required parameters are the external drivers, namely the climatic parameters: daily mean, minimum and maximum air temperature, daily mean wind speed, daily mean vapor pressure and daily total sunshine duration. From these, long- and shortwave radiation are calculated as well as the equilibrium temperature (Teq) of the water (maximum potential temperature the water could achieve after sufficient time to reach equilibrium, or water temperature at which the sum of heat fluxes across the water-air interface is 0) without shading and with full shading of the channel. (4) The model parameters are calibrated against a measurement time series of average daily water temperature for stations distributed throughout the watershed (at locations corresponding to the outflow of the sub-catchments). The calibrated parameters are (for each station) the channel shading coefficient, the tau/d ratio, a temperature offset constant (which accounts for biases due to

simplification in the soil temperature model, caused typically by elevation offsets and different albedo and slope effects); and for the entire watershed, the parameters defining the source temperature (or soil temperature Tsoil) and the leaf-area index (LAI) required for correcting the influence of shading as well as influencing the soil temperature. (5) Future water temperature can then be predicted by using future climatic data: this involves calculating a corresponding average flow and residence time in step 1, thus leading to a future prior tau/d ratio. The factor difference between the current prior tau/d and current calibrated tau/d is then applied to the future prior tau/d in order to get the future posterior tau/d. The other calibrated parameters are assumed to remain constant in future. The future Teq must also be calculated based on the future climatic series.
The technical requirements and formatting of the input data (such as resolution, land use categories, units, criteria for homogeneous sub-catchment delineation etc.) are detailed in Annexes A2.3 – A2.6.

The flow routing module [198, 199] calculates a spatially distributed simplified water balance (surface runoff, percolation and evapotranspiration based on Penman's equation) based on climatic parameters (precipitation, mean air temperature and wind speed) and physical parameters (elevation, slope, soil class and land use), from which the flow accumulation and routing, average flow and residence time are computed. The average discharge computed for Payerne for the historical climate series (1981-2011) was 7.89 m^3/s, which corresponds well with the average historical discharge (1920-2011) measured at Payerne (7.72 m^3/s).

The tau/d module calculates the spatial distribution of the temporal and flow depth components. Tau is the residence time as calculated in the first step using flow routing and climatic half-year parameters. The depth is estimated according to eq. A2.1 [200]:

$$d = 0.4Q^{0.35} \qquad \text{(A2.1)}$$

Where Q is the discharge (m^3/s) and d is the water depth (m).
Hydraulic radius according to catchment area upstream is estimated according to eq. A2.2 [201]:

$$R = 0.07A_c^{0.47} \qquad \text{(A2.2)}$$

Where R is the hydraulic radius (or flow cross-sectional area over wetted perimeter) (m) and A_c is the catchment area (km^2).

Flow velocity can then be estimated from the hydraulic radius according to the Manning-Strickler formula [202, 203] (eq. A2.3):

$$v = \frac{1}{n} R^{2/3} \sqrt{sl} \qquad \text{(A2.3)}$$

Where v is the flow velocity (m/s), n is the Manning roughness coefficient (-) and sl is the slope of the channel (-).
Finally, the residence time tau is estimated by dividing the flow velocity by the distance from the source.

The equilibrium water temperature is calculated according to Edinger et al. 1968 [204], as the water temperature for which the net heat exchange between water and air is 0 (eq. A2.4):

$$SW_{in} + LW_{in} - LW_{em} - ET - Con = 0 \qquad \text{(A2.4)}$$

Where SW_{in} is the net incoming solar radiation (with the reflected short-wave radiation already accounted for), LW_{in} is the net incoming long-wave atmospheric radiation (with the reflected long-wave radiation already accounted for), LW_{em} is the long-wave radiation emitted from the water surface (calculated with Stefan-Boltzmann's law), ET is the rate of heat loss through evaporation (essentially a function of vapor pressure, air temperature and wind speed), and Con is the rate of heat loss through conduction (calculated with the Bowen ratio, which is a function of ET, vapor pressure and air temperature) (units all in MJ/(m^2*s) for example). SW_{in} and LW_{in} (unless provided as input, as is the case of SW_{in} in the future climate series) are estimated from the sunshine duration, the vapor pressure and the mean air temperature according to the cloud cover and a day-of-the-year corrected incoming solar radiation. Cloud cover is estimated based on a regression between sunshine duration and SW_{in} for a historical time series.

The soil temperature accounts for vegetation cover, and is calculated using the differential equation from Zheng et al. (1993) [205] (eq. A2.5):

$$\frac{dT_{soil}}{dt} = M_2 \cdot (T_{air} - T_{soil}) \cdot \begin{cases} e^{(-K_{soil} \cdot LAI)} & |T_{soil} < T_{air} \\ 1 & |T_{soil} \geq T_{air} \end{cases} \qquad \text{(A2.5)}$$

Where T_{air} is the air temperature (°C), T_{soil} is the soil temperature (°C), M_2 is the rate of heat exchange between the air and the soil (s^{-1}), K_{soil} is the extinction coefficient (of radiation) due to leaf cover (-) and LAI is the one-sided leaf area index (m^2/m^2). LAI is calculated using a conceptual heat-sum model (eq. A2.6):

$$\frac{dLAI}{dt} = LAI \cdot \begin{cases} \mu_0 \cdot (T_{air} - T_0) \cdot \left(1 - \frac{LAI}{LAI_{max}}\right) & |T_{air} > T_0 \\ -k_{decay} \cdot (T_0 - T_{air}) & |T_{air} \leq T_0 \end{cases} \qquad \text{(A2.6)}$$

Where LAI_{max} is the maximum LAI attainable (according to the vegetation type) (m^2/m^2), T_o is the threshold growth temperature above which the vegetation is assumed to grow and below which it is assumed to die back (°C), μ_o is the vegetation growth rate (s^{-1}) and k_{decay} is the vegetation decay rate (s^{-1}). Furthermore, LAI cannot go below the minimum value LAI_{min}.

Assuming a steady flow stream that is well mixed along the vertical and transverse axes, the water temperature is then computed according to a one-dimensional heat transport equation, which states that the change in temperature in a stream cross-section is equal to the balance of heat import and export due to dispersion and advection, and heat exchange with the environment (eq. A2.7) [206]:

$$A_c \frac{\partial T}{\partial t} + \frac{\partial (QT)}{\partial x} = \frac{\partial}{\partial x}\left(A_c D_L \frac{\partial T}{\partial x}\right) + \frac{WS}{\rho c_p} \qquad \text{(A2.7)}$$

Where A_c is the stream cross-section area (m^2), T is the water temperature (°C), Q is the discharge (m^3/s), D_l is a dispersion coefficient in the direction of flow (m^2/s), W is the stream surface width (m), S is the net heat exchange rate with the surrounding environment ($MJ/(m^2*s)$), ρ is the density of water (kg/m^3), c_p is the specific heat of water (MJ/(kg*°C)), and x is the distance in direction of flow (m). Due to the additional assumption of plug-flow, the advection and dispersion terms can be neglected, yielding eq. A2.8:

$$\frac{dT}{dt} = \frac{S}{d\rho c_p} \qquad \text{(A2.8)}$$

Where d is the water depth (m). The net heat transfer S accounts for heat exchange at the air/water and sediment/water interfaces (eq. A2.9):

$$S = S_a + S_b \qquad \text{(A2.9)}$$

Where S_a is the net heat exchange rate between the water and the air [MJ/(m²*s)] and S_b is the net heat exchange rate between the stream bed and the water [MJ/(m²*s)]. S_a can be expressed as eq. A2.10:

$$S_a = SW_{in} + LW_{in} - LW_{em} - ET - Con \qquad \text{(A2.10)}$$

Or in terms of the equilibrium temperature, as eq. A2.11:

$$S_a = K(T_e - T) \qquad \text{(A2.11)}$$

Where K is the thermal exchange coefficient for the air-water interface (MJ/(s*m²*°C). S_b can be expressed as eq. A2.12:

$$S_b = -k\frac{\partial T_b}{\partial z} \ |z = 0 \qquad \text{(A2.12)}$$

Where k is the thermal conductivity of the stream bed, z is the vertical distance to the stream bed, and T_b is the streambed temperature, assumed identical to the soil temperature T_{soil}. This streambed or soil temperature is also assumed to be the source water temperature T_{source}, or water temperature at t=0 (as visible in eq. A2.14).

Substituting eq. 10 and eq. 11 into eq. 8, the heat transport equation becomes eq. A2.13:

$$\frac{dT}{dt} = \frac{K}{d\rho c_p}\left(T - T_{eq}\right) \qquad \text{(A2.13)}$$

The solution of eq. A2.13 yields the water temperature for a given water parcel, as given in eq. A2.14:

$$T = T_{eq} + (T_{source} - T_{eq})e^{-K\frac{\tau}{d}} \qquad \text{(A2.14)}$$

For the outlet of an entire sub-catchment, the exponential term in equation A2.14 can be well approximated by a linear function for small tau/d's. This means that the average tau/d of the sub-catchment, as given in eq. A2.15, can be used rather than having to aggregate each individual water parcel.

$$\frac{\tau}{d} = \frac{\sum_i \left(q_i \sum_j \frac{\tau_{i,j}}{d_{i,j}} \right)}{\sum_i q_i} \qquad \text{(A2.15)}$$

Where T_{source} is the soil temperature, τ is the travel time, q_i is the flow per unit width of stream for each cell i and j is the position of each water parcel from its source to the outlet of the sub-catchment, indexed by i.

Shading is accounted for in the equilibrium temperature calculation based on the share of travel time that is spent in shaded regions (estimated based on the number of channel pixels for which land use is forest), according to eq. A2.16:

$$T_{eq,eff} = sf \cdot T_{eq,shade} + (1 - sf) \cdot T_{eq} \qquad \text{(A2.16)}$$

Where sf is the shading factor (% of forested area in a sub-catchment), $T_{eq,eff}$ is the effective equilibrium temperature (°C), $T_{eq,shade}$ is the equilibrium temperature for completely shaded conditions (°C) (assuming shortwave radiation blocking of 90%), and T_{eq} is the equilibrium temperature for completely non-shaded conditions (°C).
As mentioned above, a temperature correction constant T_{offset} (°C) is added, accounting for bias in the soil temperature model and altitudinal lapse rate. Thus the water temperature at the outflow of a sub-catchment is defined by eq. A2.17:

$$T_w = T_{eq,eff} + \left(T_{source} - T_{eq,eff}\right) e^{-K\frac{\tau}{d}} + T_{offset} \qquad \text{(A2.17)}$$

A2.2. Model calibration method

The calibrated model parameters are (for each station) the shading factor sf, the tau/d ratio, the temperature offset constant T_{offset}; the parameters defining the soil temperature (M_2, K_{soil}) and the leaf-area index (μ_o, k_{decay}, LAI_{max}, LAI_{min}, T_o) were calibrated for the lower part of the catchment, separately for the channel measurement stations and the stream measurement stations (indeed, these two groups showed a different behavior; calibration for both at the same time did not yield satisfactory results). A prior distribution is defined for each of these parameters: as shown in Figure 9, the prior means of the distributions of sf and tau/d are provided by the tau/d module, whereas the prior T_{offset} is provided by the

flow routing module. For the soil and LAI parameters and the other distribution parameters of sf and tau/d, the prior distributions are based on previous studies, literature and expert estimates (these values and their sources are listed below in Table A2.8). The parameter values are then optimized so as to minimize the difference between the simulated and the observed water temperatures. The differences between the two values arises from random errors in the input data, random errors in the observation values, and errors due to non-optimal parameter values and simplifications or errors in the model structure [197]. The calibration used here allows the minimization of the last two error sources by assuming all errors arise from the model itself (while avoiding the compensation of one error type by another [207]), and that by default, all errors are independent and identically distributed (possible if there is no temporal "memory" in the errors: this is the case for flowing water, since the effect of yesterday's water temperature is no longer felt in a given cell, as the water has flowed away). The calibrated parameters are assumed to be independent of the input and state variables and are treated as random variables with a certain probability distribution. The maximum-likelihood parameter estimation method is used as the optimization criterion: this assumes that the only source of uncertainty lies in the observation values. The random errors in observation values are assumed to be independent and normally distributed. Bayesian inference is used to obtain posterior distributions of the parameters.

According to Bayes' theorem (or Bayesian inference), the posterior probability of a parameter set **P** given the observed values T_{obs}, is equal to the prior probability of the parameter set times the probability of achieving the observed values given the parameter set, weighted by the prior probability of the observed values (eq. A2.18):

$$P(\mathbf{P}|T_{obs}) = \frac{P(\mathbf{P})P(T_{obs}|\mathbf{P})}{P(T_{obs})} \qquad \text{(A2.18)}$$

The probability of the observed values is however independent of the parameter set, therefore equation A3.18 can be simplified as eq. A2.19:

$$P(\mathbf{P}|T_{obs}) \propto \mathcal{L}(T_{obs}|\mathbf{P}) \cdot P(\mathbf{P}) \qquad \text{(A2.19)}$$

Which states that the posterior probability of a parameter set **P** given the observed values T_{obs}, is proportional to likelihood of achieving the observed values given the parameter set, times the prior probability of the parameter set [208].

This can be rearranged in log form for numerical convenience, which reduces computational requirements due to smaller values, as given in eq. A2.20:

$$\log[P(\mathbf{P}|T_{obs})] \propto \log[P(\mathbf{P})] + (\log[\mathcal{L}(\mathbf{E}|\mathbf{P})]) \qquad \text{(A2.20)}$$

Where the likelihood of the observed values given the parameter set is replaced by the likelihood of the residuals **E** given the parameter set (thus making the equation independent of the absolute observed values and dependent only on the relative difference between simulated and observed values).

This constitutes the objective function of the calibration procedure. Thus the calibration procedure searches the response surface (or surface of outcomes of the objective function for the entire parameter space, i.e. all possible parameter combinations within their feasible ranges) for the maximum. For this, both local and global search methods are used (in order to avoid finding a local optimum): the global search method consists of particle swarm optimization [209], whereas the local search method consists of the Nelder-Mead approach [210].

The residuals are assumed to be normally distributed, and their likelihood is calculated according to eq. A2.21:

$$\mathcal{L}\,(E_i|\mathbf{P}) = \frac{1}{\sqrt{2\pi\sigma^2}} e^{-\frac{E_i^2}{2\sigma^2}} \qquad \text{(A2.21)}$$

Where E_i is the residual or error for observation i and σ^2 is the standard deviation of the normal distribution of the error. If the error is large, the likelihood of simulating the observed values approaches 0; if the error is small, the likelihood approaches $\frac{1}{(2\pi\sigma^2)^{0.5}}$.

Assuming the residuals are independent, the likelihood for the entire sample of observations N is calculated as the product of the likelihood for each observation (eq. A2.22):

$$\mathcal{L}(\mathbf{E}|\mathbf{P}) = \prod_{i=1}^{N} \mathcal{L}(E_i|\mathbf{P}) \qquad \text{(A2.22)}$$

In log form, this yields eq. A2.23:

$$-\log\big(\mathcal{L}(\mathbf{E}|\mathbf{P})\big) = \sum_{i=1}^{N} \left[\frac{E_i^2}{2\sigma^2} + 0.5\log(2\pi\sigma^2)\right] \qquad \text{(A2.23)}$$

In practice, the model is run with a random set of parameters and the log-likelihood is computed. This is iteratively repeated until the maximum of the objective function is achieved. Thus the corresponding parameter set is optimum parameter set for which the likelihood of simulating the observed values is the highest, given the prior probabilities of the parameters. (A non-Bayesian optimization would not account for the prior probability of the parameters: therefore, although the model fit achieved would be better than that achieved using Bayesian inference, the plausibility of the corresponding parameter set would not be considered. The use of Bayesian inference (using prior probability distributions) for the identification of the optimal parameter set deals with the problem of non-unique optima by restricting the parameter sets to probable and realistic sets [211]).

The posterior distribution of the parameter set is then sampled with the Markov Chain Monte Carlo (MCMC) technique, appropriate for multi-parameter models with correlated parameters [212, 213]. This relies on the fact that the equation of the posterior distribution cannot be calculated, but the value of the probability can be calculated for any parameter combination. In practice the parameters are randomly sampled within their distributions, and the resulting probability values are retained or discarded depending on whether they are better or only slightly worse (in which case they are kept with a certain fixed probability) than the probability value calculated for the previous sample. The resulting number of calculated probabilities retained along the entire range converge to a proportion of a true statistical distribution, providing the shape of the posterior distribution.

A2.3. Flow routing module: input data and assumptions

The half years range from May to October and from November to April. All raster inputs need a consistent resolution (e.g. 25 m*25 m). Land use should be assigned a code according to the categories shown in Table A2.1; soil class should be assigned a code according to Table A2.2, and slope should be assigned a code according to Table A2.3.

Table A2.1: *Land use categories used.*

Code	Land use category in model
1	City center
2	Residential area
3	Sparse buildings
4	Transportation
5	Airport

6	Industrial areas
7	Excavation sites
8	Arable crops
9	Meadow
10	Orchard
11	Deciduous forest
12	Coniferous forest
13	Mixed forest
14	Shrubland
15	Sparse vegetation
16	Mud flat/marsh
17	Open water

Table A2.2: *Soil classes used [196, 214].*

Code	Soil class in model	Saturated hydraulic conductivity (µm/s)
1	Sand	>42.34
2	Sandy loam	14.11-42.34
3	Loam	4.23-14.11
4	Clay loam	1.41-4.23
5	Clay	0.42-1.41
6	Peat	-
7	Bare rock	0-0.42

Table A2.3: *Slope classes used.*

Code	Slope interval
1	0-1
2	1-2
3	2-7
4	7-12
5	12-18
6	18-24
7	24-150

The sub-catchments are delineated following the topography and stream network. Their size limit should ideally be roughly between 0 and 30 km^2 in order to respect the conditions necessary for the model assumption that tau/d can be averaged within a sub-catchment only if tau/d is generally small. The setup for the Broye violates this assumption at the lower observation stations, however the ensuing error can be compensated for through calibration.The measurements are then taken at a selection of these sub-catchments, representing the range of occurring hydrological response units (HRU's, see below) and of stream orders, in accessible locations and at a minimal distance from intersections and perturbing factors (e.g. waste-water treatment plant outflows).

The HRU's are provided by a classification of the whole watershed into units which have homogeneous rainfall-runoff characteristics (i.e. land use, saturated hydraulic conductivity of the soil, and elevation). This simplifies the spatial variability in the basin, while accounting for the differences in the most important factors for the runoff estimation. The classification is given in Table A2.4:

Table A2.4: *Hydrological response unit classification.*

HRU number	Land use	Elevation (masl)	Saturated hydraulic conductivity (µm/s)
HRU1	Urban	-	-
HRU2	Agricultural	≥ 800	0.42-1.41
HRU3			1.41-42.34
HRU4		< 800	0.42-1.41
HRU5			1.41-42.34
HRU6	Forest	≥ 800	1.41-42.34
HRU7			0.42-1.41
HRU8		< 800	1.41-42.34
HRU9			0.42-1.41

The units of the other inputs for the flow routing module are provided in Table A2.5:

Table A2.5: *Units of climatological inputs for flow routing.*

Parameter	Unit
Precipitation, half year sum	mm
Wind speed, half year average	m/s

Air temperature, half year average	°C

Interpolated air temperature must account for the altitude lapse rate, for example using equation A2.24:

$$T_z = T_r - (z - z_r) * lr \qquad \text{(A2.24)}$$

Where T_z is the altitude-corrected temperature (°C) at altitude z (m), T_r (°C) is the temperature at the reference altitude z_r (m), and lr is the lapse rate (°C/m altitude), which can range from 0.5 to 1 °C/100 m according to the air humidity. The internationally recommended average of 0.64 °C/100 m was used here.

The air temperature must be converted to the temperature-dependent term of the Penman evapotranspiration equation, in order to be integrated in the flow routing module, as given by eq. A2.25:

$$Penman(T) = 1.5411 * e^{-0.057*T_z} \qquad \text{(A2.25)}$$

The climatological time-series from 1981 to 2011 for the weather stations Avenches, Payerne, Oron, Moudon, and Semsales were used to calculate the historical precipitation, temperature and wind speed, by interpolation from the averaged time-series data using the Thiessen polygons method, in order to provide continuous climate values for the entire watershed.

A2.4. Teq module: input data and assumptions

For Teq, only the climate station Payerne was used: this was assumed to be representative for the temperature fluctuations of all the water temperature stations in the lower catchment. The units of the input parameters are given in Table A2.6.

Table A2.6: *Units of climatological inputs for Teq.*

Parameter	Unit
Air temperatures	°C
Wind speed at 2m above ground	m/s
Vapor pressure	hPa
Sun duration	h
Gross incoming solar radiation	W/m^2

A2.5. Tau/d module: input data and assumptions

For this study, Manning's roughness coefficient was assumed to be 0.2 (a typical value for earth channels or streams).

The means of the prior distributions of tau/d, sf and T_{offset} are calculated in the tau/d module. The other prior distribution parameters assumptions are given in Table A2.7.

Table A2.7: *prior distributions of tau/d, sf and T_{offset} [196, 214].*

Parameter	Distribution	Std dev	Global min	Global max
tau/d	lognormal	0.5	0	2
sf	beta	0.2	0	1
T_{offset}	normal	2	-10	5

A2.6. Tsoil module: input data and assumptions

The prior distributions for the Tsoil module are given in Table A2.8.

Table A2.8: *prior distributions for the Tsoil module [108, 196].*

Parameter	Distribution	Mean	Std dev	Global min	Global max	Alpha	Beta
K_{soil}	beta	0.2	0.2	0	1	5	5
M_2	beta	0.025	0.2	0	1	4.4	13.3
T_o	normal	10	0.3	5	15		
μ_o	lognormal	0.02	0.002	0	0.1		
k_{decay}	lognormal	0.02	0.002	0	0.1		
LAI_{min}	lognormal	1	0.1	0	2		
LAI_{max}	lognormal	4	0.5	1	6		

The beta distribution is used for K_{soil} and M_2 because these parameters fall between 0 and 1, but the probability of being be equal to 0 or 1 is nil. Lognormal distributions are used in order to exclude negative numbers based on theoretical restrictions, whereas the parameters with normal distributions can include negative numbers. These parameters are based on previous water temperature modeling studies in two test catchments in Switzerland (the Gürbe (117 km^2) and the Mönchaltofer Aa (46 km^2), realized in the context of the iWaQa project. The choice of a realistic parameter range should produce a

parameter space that is wide enough to include all possible parameter combinations, while avoiding parameters that have no meaning [211]. The standard deviation of the LAI parameters had to be reduced in the current case in order to obtain realistic predictions of LAI.

A2.7. Twater module: input data and assumptions

The measurements for calibration were taken using Onset HOBO Pendant® loggers (serial number UA-002-08), with a measurement range of -20°C to 70°C, an accuracy of ± 0.53°C, a resolution of 0.14°C at 2°C and a response time in water of 5 min.

The depth-specific heat exchange coefficient K (used in eq. A3.14) is estimated from the climate data, using a site-independent multivariate linear regression model developed in the iWaQa project. K can be calculated empirically from climate data [215], however using the regression model saves computation time. The value obtained using the regression value is very close to the empirically calculated value, and shows only small variations in any case.

A2.8. Calibration and validation results

The calibrated parameter values for the channel and stream stations in the Broye catchment are provided in Tables A2.9 and A2.10 respectively.

Table A2.9: *Calibrated parameter values for channel station calibration (individual and common parameters)*

Parameter	**tau/d**	**sf**	**T_{offset}**	**K_{soil}**	**M_2**	**T_o**	**μ_o**	**k_{decay}**	**LAI_{min}**	**LAI_{max}**
Unit	s/m	-	°C	-	s^{-1}	°C	s^{-1}	s^{-1}	-	-
Station	Calibrated value									
1	0.83	0.11	-0.22							
3	1.54	0.06	-0.22							
5	1.13	0.12	-0.27							
6	1.12	0.14	-0.24							
10	1.12	0.12	-0.17	0.027	0.198	10.6	0.019	0.023	1.08	5.07
11	1.10	0.12	-0.25							
15	1.28	0.15	-0.37							
16	0.81	0.20	-0.41							
17	0.90	0.24	-0.60							

Table A2.10: *Calibrated parameter values for stream station calibration (individual and common parameters)*

Parameter	**tau/d**	**sf**	**T_{offset}**	**K_{soil}**	**M_2**	**T_o**	**μ_o**	**k_{decay}**	**LAI_{min}**	**LAI_{max}**
Unit	s/m	-	°C	-	s^{-1}	°C	s^{-1}	s^{-1}	-	-
Station	Calibrated value									
2	1.41	0.01	0.69							
4	1.26	0.04	0.61							
7	0.82	0.33	0.13							
8	0.77	0.19	0.24							
9	0.62	0.20	-0.21							
12	0.46	0.31	-0.54	0.32	0.12	10.22	0.016	0.020	0.97	8.04
13	0.49	0.26	-0.55							
14	0.40	0.22	-0.96							
18	0.60	0.20	-1.34							
19	0.54	0.44	-0.39							
20	0.54	0.15	-0.12							

K_{soil} is the parameter which differs most between the channel and stream stations. The very low value of K_{soil} for the channel stations indicates that the soil temperature model is behaving in a unexpected way in order to allow the modeled water temperature to achieve a good fit. This suggests that the model is conceptually too simplified for the case of the channel stations, in particular that an additional source of heat has not been accounted for. This could be wastewater treatment plants for example, or an interaction with a thick substrate layer with different heat properties. The model is nevertheless able to predict the water temperature in a satisfactory way (although predictions of soil temperature themselves cannot be considered reliable for the channel sub-catchments: indeed, this parameter must now be seen to include other influencing factors and not only soil temperature). This is further shown by the excessive variability of soil temperature in Figure A2.2. An example of a poor fit is shown in Figure A2.1.

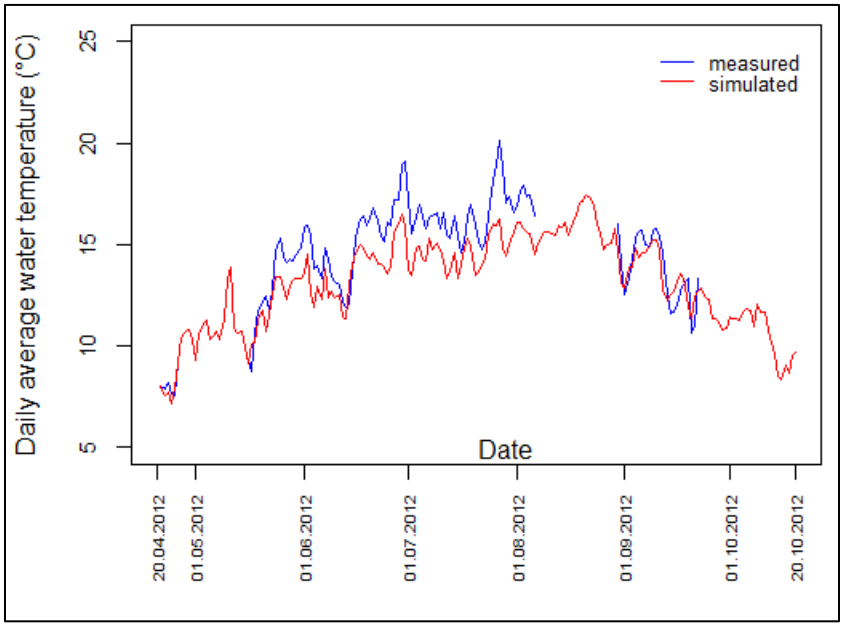


Figure A2.1: *measurements and simulations for station 18 (with poor model performance and short measurement series).*

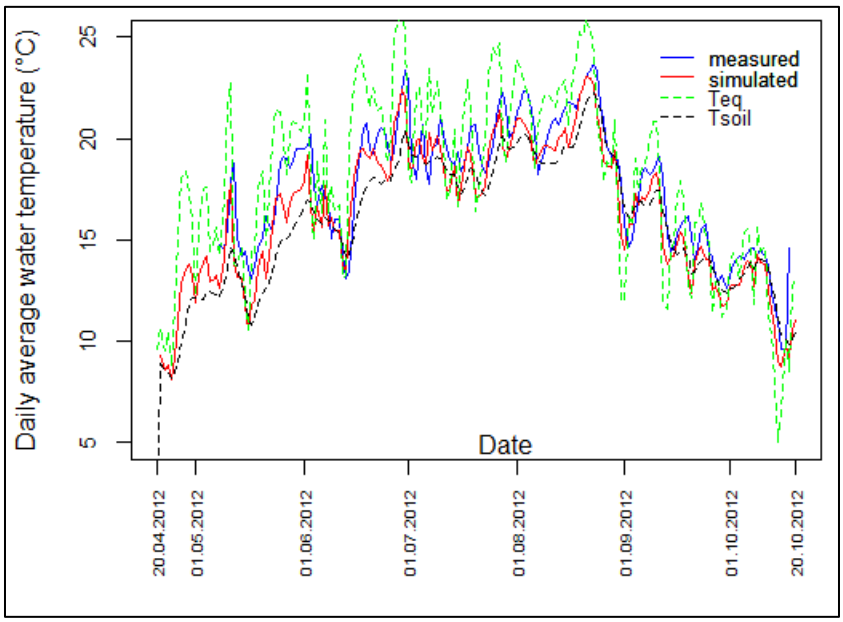


Figure A2.2: *measurements and simulations for station 1, with two key parameters (soil temperature Tsoil and equilibrium temperature Teq).*

The water temperature typically varies between soil temperature Tsoil, and equilibrium temperature Teq; in summer, water temperature should be below Teq (due to cooler groundwater inflows), whereas

in autumn and winter, this may be reversed (groundwater inflows maintain a relatively constant temperature, and therefore in winter can be warmer than external temperatures).

Tsoil is quite variable for the channel station calibration (shown for the example of station 1): this reflects the issue of an over-simplified soil temperature model discussed above.

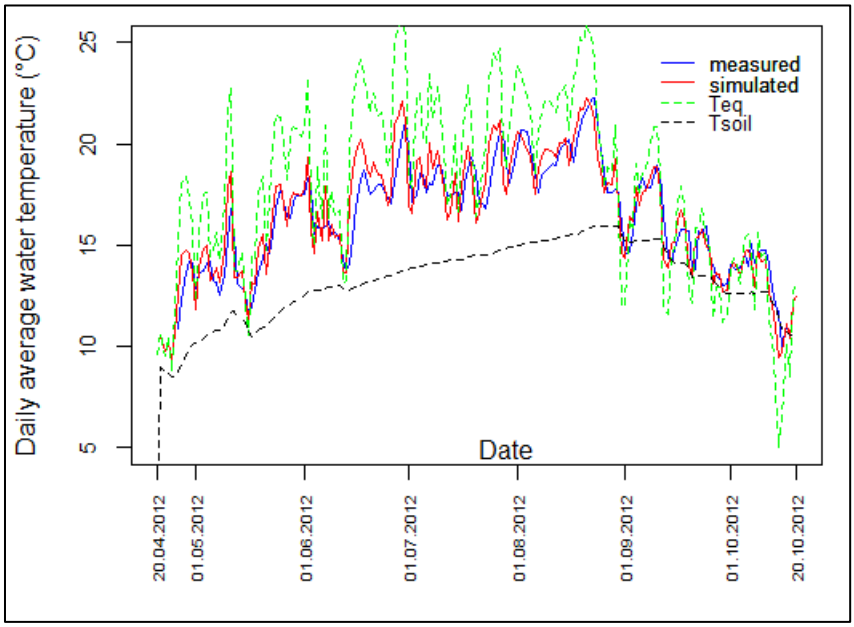


Figure A2.3 : *measurements and simulations for station 2, with two key parameters (soil temperature Tsoil and equilibrium temperature Teq).*

For the stream station calibration (station 2 is shown as an example in Figure A2.3), Tsoil behaves in a predictable way, with low variability and temperatures which are consistently lower than water temperature in summer.

ANNEX 3: LIFE CYCLE ASSESSMENT OF FARM ADAPTATION SCENARIOS

A3.1. Description of farm optimization model

The two farm types available for this study consist of a modeled 30 ha arable crop farm [7], respectively mixed farm (arable crops and livestock) [216]. These modeled farms represent average farms for their type in the region, as identified by a statistical analysis of farm accountancy data. Their environmental characteristics (local climate, soil and slope) were taken for a representative location in the lowland area of the respective regions.

The economic farm optimization is depicted in Figure A3.1 (adapted from Lehmann (2013) [7]).

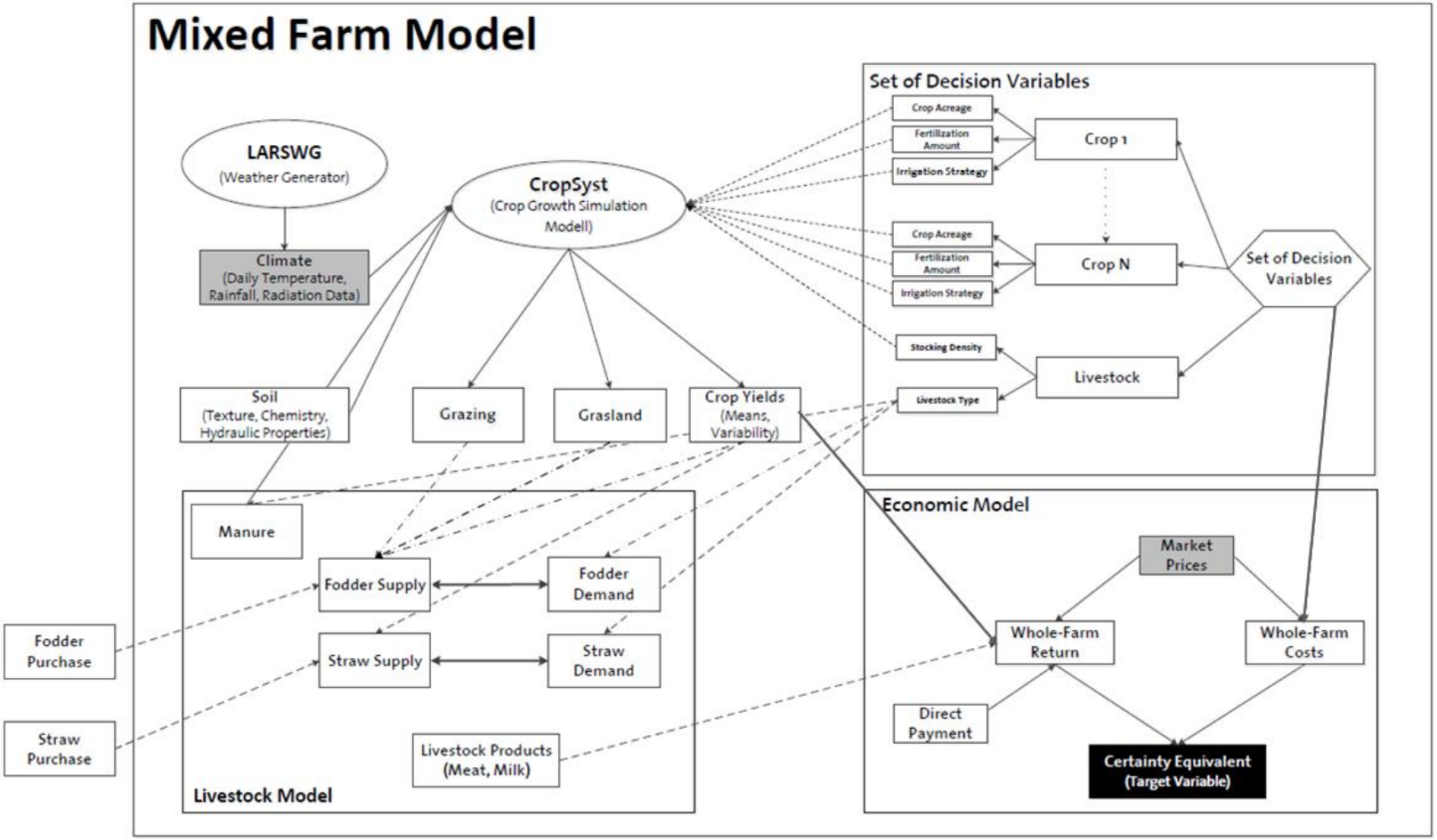


Figure A3.1: *Mixed farm optimization model from Lehmann (2013) [7]. Rectangles = data, ovals = external models, hexagons = databases; grey fill = scenario data, black fill = optimization objective; large-dashed lines = only for mixed farms (livestock-related flows), small-dashed lines = optional flows (depending on set of variables used), dot-dashed lines = conditional flows (constraints on fodder production based on livestock quantities).*

It uses the crop growth model CropSyst [217] to simulate crop yields based on management variables (crop mix and acreage, fertilization amount, irrigation intensity, livestock type and number), and environmental conditions (climate, soil type). Climate input data is downscaled to daily weather data using the weather generator LARS WG [218]. The mixed farm model also contains a simple livestock model (related to the crop model via pasture, grass and silo maize requirements, as well as manure application) [216]. Outputs are modeled for a 25 year simulation period. The optimization goal is the maximization of farm gross profit margin (i.e. profit before considering salaries and capital costs such as infrastructure) while minimizing its volatility. This goal is measured by the certainty equivalent, a combined indicator of profit (defined as net income before salary and capital cost deduction) and its inter-annual variability [7], which includes the farmers aversion to risk, and accounts for product prices and input costs. The optimization variables of the model correspond to the primary constituents of farm management: crop choice and surface allocation, animal type, number and stocking density (for the mixed farm), and for each crop: nitrogen fertilization intensity and irrigation intensity. The crops considered are winter wheat, winter barley, grain maize, rapeseed, potato and sugar beet, as well as silo maize, grassland and pasture for mixed farms. Crop rotations were not considered. The livestock types considered are dairy cows, nursing cows, calves for fattening, bulls for fattening and bulls for breeding. Pest control strategy, irrigation technology or tillage strategy are not considered as variables in the model, due to excessive complexity in the modeling requirements and less robust responses of the crop growth model to these parameters.

A3.2. Price change scenario

The prices assumed for the price change scenario are listed in Table A3.1.

Table A3.1: *product prices assumed in the farm optimization model for the reference scenario (current Swiss prices) and the European price scenario (current Austrian prices). Source: Lehmann 2013 [7].*

Crop	Average Swiss price (2002-2009)	Average Austrian price (2002-2009)
Wheat	51.4 CHF/dt	24.3 CHF/dt
Barley	37.9 CHF/dt	16.7 CHF/dt
Rapeseed	78.7 CHF/dt	43.4 CHF/dt
Grain maize	37.9 CHF/dt	20.0 CHF/dt
Silo maize	3.8 CHF/dt	2.0 CHF/dt
Sugar beet	5.4 CHF/dt	4.9 CHF/dt

Hay	34.7 CHF/dt	17.3 CHF/dt
Milk	70.5 CHF/l	48.9 CHF/l
Bulls MT H3	8.7 CHF/kg dead weight	4.7 CHF/kg dead weight
Calves KV T3	13.35 CHF/kg dead weight	7.5 CHF/kg dead weight
Natura Beef T3	10.44 CHF/kg dead weight	7.5 CHF/kg dead weight
Straw	16.6 CHF/dt	9.8 CHF/dt

A3.3. Subsidy change scenario

The subsidies assumed in the subsidy change scenario are listed in Table A3.2.

Table A3.2: *subsidies assumed in the farm optimization model for the reference scenario (current subsidy levels) and the subsidy change scenario (future levels as foreseen in ongoing policy debates). Source: Lehmann 2013 [148].*

Subsidy type	Reference scenario	Subsidy change scenario
Area contributions	1040 CHF/ha	850 CHF/ha
Additional contributions for arable soils and permanent crops	640 CHF/ha	200 CHF/ha
Contributions for sugar beet for sugar production	1900 CHF/ha	1500 CHF/ha
Contributions for oil crops (rapeseed, sunflower, oil pumpkins and linseed) and soja	1000 CHF/ha	800 CHF/ha
Contributions for bovine livestock	690 CHF/livestock unit	0 CHF/ livestock unit
Contributions for dairy cows	450 CHF/ livestock unit	0 CHF/ livestock unit
Contributions for regular access to outdoors for livestock	180 CHF/ livestock unit	180 CHF/ livestock unit
Contributions for livestock housing with particularly high living quality	90 CHF/ livestock unit	90 CHF/ livestock unit

A3.4. Absolute impacts and functional units of all scenarios

The absolute impacts (not related to a functional unit) as well as the functional units for all farm scale scenario are listed in Table A3.3.

Table A3.3: *absolute impacts (for selected impact categories) and the three functional units, for all scenarios assessed at the farm scale.*

Farm type	Region	Scenario	GWP (10^3 kg CO_2 eq.)	ABL (GSE*y)	TEP (1,4-DB eq.)	Total production (10^3 MJ dig. en.)	Crop production (10^3 MJ dig. en.)	Animal production (10^3 MJ dig. en.)	Profit (10^3 CHF)	Cultivated area-time (ha*y)
Mixed	Broye	Reference	192.37	1.03E-09	294.01	1634.51	931.4	703.1	197.88	30
		Ext2050 without adaptation	183.51	2.11E-09	287.42	1486.25	822.3	664.0	183.90	30
		Ext2050	194.09	8.28E-09	288.46	1717.88	1014.8	703.1	190.88	30
		Mod2050	189.40	2.02E-09	280.80	1442.20	719.6	722.6	102.81	30
		Ext2050, subsidy change	196.34	8.28E-09	300.20	1697.27	994.2	703.1	164.37	30
		Ext2050, European prices	177.95	2.29E-09	230.67	1159.35	436.7	722.6	138.45	30
		Ext2050, water pricing	187.49	1.91E-09	279.96	1460.07	757.0	703.1	186.42	30
		Ext2050, water quota	180.83	0	269.34	1342.71	639.6	703.1	187.05	30
	Greifensee	Reference	194.22	0	284.56	2109.44	1406.4	703.1	209.54	30
		Ext2050 without adaptation	187.11	0	280.59	1608.84	925.3	683.6	188.94	30
		Ext2050	189.31	1.92E-09	279.21	1448.15	725.5	722.6	194.06	30
		Mod2050	193.92	0	284.32	2001.92	1298.8	703.1	200.96	30
		Ext2050, subsidy change	188.30	0	250.03	1894.07	1171.5	722.6	164.43	30
		Ext2050, European prices	183.70	0	229.80	1170.88	428.7	742.1	146.64	30
Crop	Broye	Reference	106.59	6.28E-09	391.67	2954.35	2954.4	0	116.50	30
		Ext2050 without adaptation	101.04	1.03E-08	390.61	2619.57	2619.6	0	100.17	30
		Ext2050	93.34	1.37E-08	382.82	2593.40	2593.4	0	101.18	30
		Mod2050	107.58	1.10E-08	385.25	2745.41	2745.4	0	104.77	30
		Ext2050, subsidy change	99.59	1.52E-08	378.01	2684.49	2684.5	0	78.51	30
		Ext2050, European prices	69.27	1.13E-08	357.22	2192.77	2192.8	0	47.43	30

	Ext2050, water pricing	84.74	1.79E-09	381.44	2191.81	2191.8	0	92.55	30
	Ext2050, water	84.09	2.05E-09	378.27	2155.32	2155.3	0	94.76	30
Greifensee	Reference	101.33	1.50E-09	379.43	2784.56	2784.6	0	116.23	30
	Ext2050 without adaptation	91.90	1.76E-09	377.70	2197.27	2197.3	0	91.97	30
	Ext2050	78.07	9.16E-09	375.21	2309.93	2309.9	0	94.25	30
	Mod2050	100.49	1.46E-09	361.75	2203.15	2203.2	0	100.03	30
	Ext2050, subsidy change	77.91	9.16E-09	375.18	2309.63	2309.6	0	70.85	30
	Ext2050, European prices	57.29	6.87E-09	355.28	1962.88	1962.9	0	46.47	30
	Ext2050, water pricing	74.02	1.76E-09	374.69	2155.19	2155.2	0	92.10	30
	Ext2050, water	74.02	1.76E-09	374.69	2155.19	2155.2	0	93.95	30

*This scenario has identical outcomes as "Ext2050, water pricing", except profit which increases slightly. The optimal management is nevertheless the same in both scenarios.

A3.5. Yields

The yields of the farm scale scenarios assessed are shown in Figure A3.2.

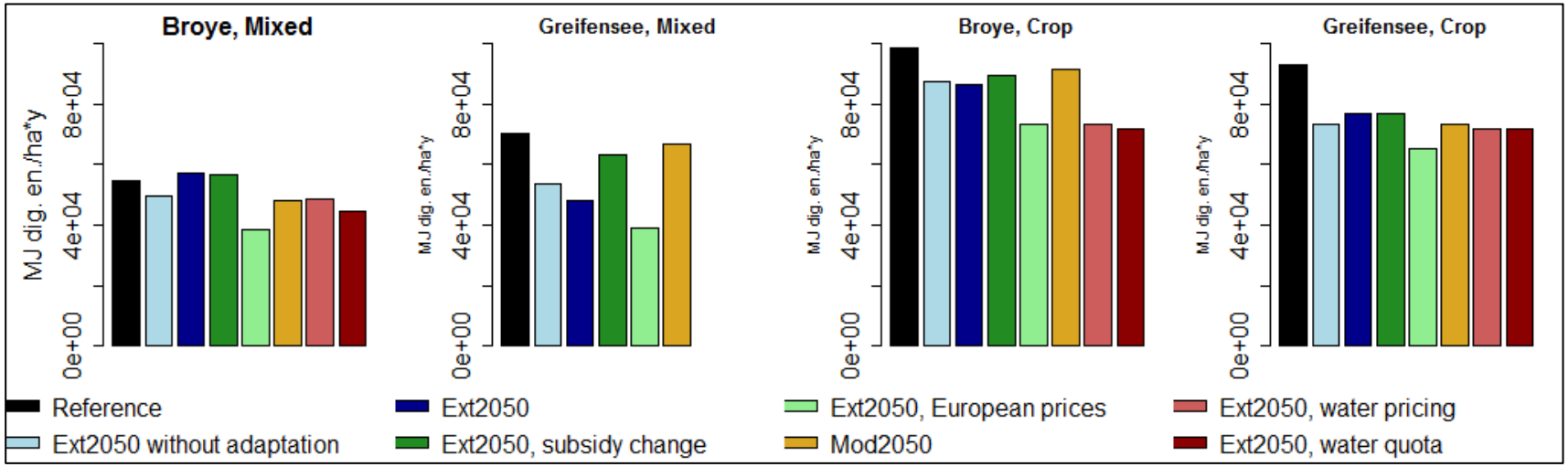


Figure A3.2: *yields of all scenarios, as MJ dig. en. per ha*y, grouped per region and per farm type.*

A3.6. Correlation matrix of initial impact categories

The correlation between the impact categories assessed at the farm scale is given in Table A3.4.

Table A3.4: *correlation between impact categories assessed at the farm scale (initial selection), with functional unit MJ dig. en.*

	Non-renewable energy	GWP	Ozone formation	Acidification	Terrestrial Eutrophication	Marine eutrophication	Freshwater eutrophication	Human toxicity	Terrestrial Ecotoxicity	Aquatic ecotoxicity	Land competition	Aquatic biodiversity loss	Terrestrial biodiversity reduction
Non-renewable energy	1.00												
GWP	0.98	1.00											
Ozone formation	0.99	1.00	1.00										
Acidification	0.96	1.00	0.99	1.00									
Terrestrial Eutrophication	0.95	0.99	0.98	1.00	1.00								
Marine eutrophication	0.93	0.94	0.94	0.93	0.93	1.00							
Freshwater eutrophication	0.98	0.99	0.99	0.98	0.98	0.97	1.00						
Human toxicity	1.00	0.98	0.99	0.96	0.95	0.93	0.98	1.00					
Terrestrial Ecotoxicity	0.72	0.60	0.65	0.54	0.52	0.55	0.61	0.72	1.00				
Aquatic ecotoxicity	0.99	0.98	0.99	0.96	0.95	0.94	0.98	0.99	0.72	1.00			
Land competition	0.94	0.97	0.97	0.97	0.97	0.88	0.95	0.95	0.63	0.94	1.00		
Aquatic biodiversity loss	-0.41	-0.46	-0.43	-0.48	-0.48	-0.46	-0.49	-0.42	-0.03	-0.37	-0.44	1.00	
Terrestrial biodiversity reduction	0.92	0.94	0.94	0.94	0.94	0.83	0.92	0.93	0.66	0.91	0.99	-0.43	1.00

A3.7. Absolute impacts for all initial impact categories

The absolute impacts (not related to a functional unit) for all the impact categories assessed and for all scenarios are listed in Table A3.5 for mixed farms and Table A3.6 for crop farms.

Farm type	Region	Scenario	Non-renewable energy	GWP	Ozone formation	Acidification	Terrestrial Eutrophication	Marine eutrophication	Freshwater eutrophication	Human toxicity	Terrestrial Ecotoxicity	Aquatic ecotoxicity	Land competition	Aquatic biodiversity loss	Terrestrial biodiversity reduction
		Unit	MJ eq.	kg CO_2 eq.	m^2*ppm*h	m^2	m^2	kg P eq.	kg N eq.	kg 1,4-DB eq			m^2*y	GSE*y	points lost
Mixed	Broye	Ref	1.43E+06	1.92E+05	1.55E+06	3.67E+04	1.38E+05	1.83E+03	4.38E+01	6.99E+04	4.31E+00	3.18E+04	4.39E+05	1.03E-09	1.05E+01
		Ext2050 without adaptation	1.38E+06	1.84E+05	1.49E+06	3.46E+04	1.30E+05	1.75E+03	4.20E+01	6.77E+04	4.29E+00	3.08E+04	4.31E+05	2.11E-09	1.05E+01
		Ext2050	1.46E+06	1.94E+05	1.56E+06	3.66E+04	1.38E+05	1.79E+03	4.50E+01	7.00E+04	4.27E+00	3.27E+04	4.39E+05	8.28E-09	1.05E+01
		Mod2050	1.41E+06	1.89E+05	1.54E+06	3.65E+04	1.38E+05	1.82E+03	4.44E+01	6.93E+04	4.17E+00	3.13E+04	4.37E+05	2.02E-09	1.06E+01
		Ext2050, subsidy change	1.48E+06	1.96E+05	1.58E+06	3.69E+04	1.39E+05	1.79E+03	4.49E+01	7.07E+04	4.36E+00	3.32E+04	4.40E+05	8.28E-09	1.04E+01
		Ext2050, European prices	1.33E+06	1.78E+05	1.46E+06	3.39E+04	1.28E+05	1.83E+03	4.36E+01	6.56E+04	3.97E+00	2.70E+04	4.40E+05	2.29E-09	1.08E+01
		Ext2050, water pricing	1.40E+06	1.87E+05	1.53E+06	3.60E+04	1.35E+05	1.79E+03	4.49E+01	6.90E+04	4.34E+00	3.12E+04	4.38E+05	1.91E-09	1.05E+01
		Ext2050, water quota	1.37E+06	1.81E+05	1.50E+06	3.57E+04	1.35E+05	9.12E+02	3.79E+01	6.84E+04	4.14E+00	3.04E+04	4.38E+05	0.00E+00	1.06E+01
	Greifensee	Ref	1.43E+06	1.94E+05	1.56E+06	3.67E+04	1.38E+05	1.90E+03	4.71E+01	7.18E+04	4.14E+00	3.08E+04	4.38E+05	0.00E+00	1.07E+01
		Ext2050 without adaptation	1.39E+06	1.87E+05	1.52E+06	3.54E+04	1.33E+05	1.85E+03	4.59E+01	7.01E+04	4.20E+00	3.01E+04	4.32E+05	0.00E+00	1.06E+01
		Ext2050	1.40E+06	1.89E+05	1.54E+06	3.62E+04	1.36E+05	1.86E+03	4.71E+01	7.00E+04	4.19E+00	3.02E+04	4.40E+05	1.92E-09	1.06E+01
		Mod2050	1.43E+06	1.94E+05	1.56E+06	3.65E+04	1.37E+05	1.91E+03	4.85E+01	7.15E+04	4.12E+00	2.87E+04	4.38E+05	0.00E+00	1.07E+01
		Ext2050, subsidy change	1.39E+06	1.88E+05	1.51E+06	3.63E+04	1.37E+05	1.82E+03	4.77E+01	6.88E+04	4.04E+00	3.01E+04	4.40E+05	0.00E+00	1.07E+01
		Ext2050, European prices	1.33E+06	1.84E+05	1.47E+06	3.56E+04	1.35E+05	1.89E+03	4.59E+01	6.68E+04	3.88E+00	2.77E+04	4.46E+05	0.00E+00	1.09E+01
		Ext2050, water pricing													
		Ext2050, water quota													

Table A3.5: *absolute impacts for all impact categories (initial selection), for all scenarios and both regions, mixed farms only. This represents the total impacts, without relation to a functional unit.*

Farm type	Region	Scenario	Non-renewable energy	GWP	Ozone formation	Acidification	Terrestrial Eutrophication	Marine eutrophication	Freshwater eutrophication	Human toxicity	Terrestrial Ecotoxicity	Aquatic ecotoxicity	Land competition	Aquatic biodiversity loss	Terrestrial biodiversity reduction
		Unit	MJ eq.	kg CO_2 eq.	m^2*ppm*h	m^2	m^2	kg P eq.	kg N eq.	kg 1,4-DB eq			m^2*y	GSE*y	points lost
Crop	Broye	Ref	9.05E+05	1.07E+05	7.63E+05	1.14E+04	3.75E+04	9.45E+02	2.25E+01	4.65E+04	5.04E+00	2.25E+04	3.08E+05	6.28E-09	9.75E+00
		Ext2050 without adaptation	8.77E+05	1.01E+05	7.48E+05	1.06E+04	3.45E+04	9.58E+02	2.20E+01	4.56E+04	5.04E+00	2.22E+04	3.07E+05	1.03E-08	9.75E+00
		Ext2050	8.44E+05	9.33E+04	7.28E+05	9.56E+03	3.04E+04	8.35E+02	2.13E+01	4.42E+04	5.22E+00	2.04E+04	3.08E+05	1.37E-08	9.57E+00
		Mod2050	9.78E+05	1.08E+05	7.70E+05	1.11E+04	3.58E+04	9.54E+02	2.42E+01	4.72E+04	4.93E+00	2.28E+04	3.11E+05	1.10E-08	9.86E+00
		Ext2050, subsidy change	9.38E+05	9.96E+04	7.50E+05	9.94E+03	3.13E+04	9.37E+02	2.35E+01	4.58E+04	5.08E+00	2.20E+04	3.11E+05	1.52E-08	9.71E+00
		Ext2050, European prices	7.00E+05	6.93E+04	6.39E+05	6.47E+03	1.90E+04	7.67E+02	1.48E+01	3.69E+04	5.52E+00	1.69E+04	3.07E+05	1.13E-08	9.27E+00
		Ext2050, water pricing	7.80E+05	8.47E+04	6.92E+05	8.65E+03	2.71E+04	8.20E+02	2.04E+01	4.29E+04	5.26E+00	1.92E+04	3.07E+05	1.79E-09	9.53E+00
		Ext2050, water quota	7.74E+05	8.41E+04	6.87E+05	8.58E+03	2.69E+04	8.14E+02	1.99E+01	4.23E+04	5.28E+00	1.88E+04	3.07E+05	2.05E-09	9.51E+00
	Greifensee	Ref	9.26E+05	1.01E+05	7.39E+05	1.05E+04	3.38E+04	8.83E+02	2.40E+01	4.60E+04	4.82E+00	2.03E+04	3.10E+05	1.50E-09	9.97E+00
		Ext2050 without adaptation	8.80E+05	9.19E+04	7.13E+05	9.26E+03	2.90E+04	8.31E+02	2.28E+01	4.45E+04	5.11E+00	1.98E+04	3.10E+05	1.76E-09	9.68E+00
		Ext2050	8.34E+05	7.81E+04	6.85E+05	7.20E+03	2.06E+04	8.10E+02	2.16E+01	4.22E+04	5.25E+00	1.97E+04	3.10E+05	9.16E-09	9.54E+00
		Mod2050	1.15E+06	1.00E+05	7.36E+05	9.20E+03	2.70E+04	9.29E+02	2.90E+01	4.74E+04	4.90E+00	2.08E+04	3.20E+05	1.46E-09	9.89E+00
		Ext2050, subsidy change	8.33E+05	7.79E+04	6.84E+05	7.18E+03	2.05E+04	8.10E+02	2.15E+01	4.22E+04	5.26E+00	1.97E+04	3.10E+05	9.16E-09	9.53E+00
		Ext2050, European prices	6.41E+05	5.73E+04	6.05E+05	4.90E+03	1.28E+04	7.22E+02	1.37E+01	3.50E+04	5.52E+00	1.62E+04	3.07E+05	6.87E-09	9.27E+00
		Ext2050, water pricing	7.92E+05	7.40E+04	6.62E+05	6.89E+03	1.97E+04	8.05E+02	2.11E+01	4.16E+04	5.26E+00	1.87E+04	3.10E+05	1.76E-09	9.53E+00
		Ext2050, water quota	7.92E+05	7.40E+04	6.62E+05	6.89E+03	1.97E+04	8.05E+02	2.11E+01	4.16E+04	5.26E+00	1.87E+04	3.10E+05	1.76E-09	9.53E+00

Table A3.6: *absolute impacts for all impact categories (initial selection), for all scenarios and both regions, crop farms only. This represents the total impacts, without relation to a functional unit.*

A3.8. Inventories of farm scale scenarios

Table A3.7 provides the inventories of all mixed farm scenarios for the Broye region; Table A3.8 for mixed farms in the Greifensee region; Table A3.9 for crop farms in the Broye region; Table A3.10 for crop farms in the Greifensee region. All scenarios are for 30 ha of land cultivated during one year.

Table A3.7: *inventories of all mixed farm scenarios for the Broye region, according to ecoinvent and SALCA nomenclature (continued pages 162 to 165).*

		Scenario							
Flow	Unit	Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Resources									
Water, river	l	1837500	3780000	14822000	14822000	4095000	3622500	3420000	
Materials/fuels									
[sulfonyl]urea-compounds, at regional storehouse/CH S	kg	3.6215	3.6215	3.5232	3.8156	0.28684	3.2315	3.3285	2.8437
[thio]carbamate-compounds, at regional storehouse/CH S	kg	1.1699	1.1748	1.1614	1.1841	1.6663	1.1308	1.167	1.0915
2,4-D, at regional storehouse/CH S	kg	0.96305	0.96394	0.93335	0.99268	0.21675	0.88233	0.8976	0.8308
Acetamide-anillide-compounds, at regional storehouse/CH S	kg	0.60364	0.60364	0.51741	0.51741	3.6772	0.60364	0.51741	0.86235
Agricultural machinery, general, production/CH/I S	kg	485.28	487.52	472	492.18	269.34	457.32	469.79	434.59
Agricultural machinery, tillage, production/CH/I S	kg	130.82	131.71	125.4	133.23	100.26	120.51	122.47	115.78
Ammonium nitrate, as N, at regional storehouse/RER S	kg	1707	1593.9	1658.5	1720.6	1142.9	1512.4	1529.7	1495.8
Ammonium sulphate, as N, at regional storehouse/RER S	kg	131.31	122.61	127.58	132.35	87.915	116.34	117.67	115.06
Atrazine, at regional storehouse/CH S	kg	0.27363	0.27363	0.23454	0.23454	0.27363	0.27363	0.23454	0.3909
Baling/CH S	p	0.096677	0.096677	0.096677	0.1289	0.032226	0.096677	0.096677	0.096677
Benzimidazole-compounds, at regional storehouse/CH S	kg	3.9733	3.9733	3.8659	4.188		3.5437	3.6511	3.1142
Benzo[thia]diazole-compounds, at regional storehouse/CH S	kg	2.4168	2.4168	2.3196	2.4933	0.27468	2.1852	2.2039	2.0713
Bucket milking system, at farm/p/CH/I U	p	1.1009	1.0397	1.1009	1.1009	1.1314	1.1314	1.1009	1.1009
Calcium ammonium nitrate, as N, at regional storehouse/RER S	kg	853.49	796.95	829.25	860.29	571.45	756.22	764.86	747.88
Cattle for breeding, Swiss integrated production, at farm gate/CH U	kg	5940	5610	5940	5940	6105	6105	5940	5940
Combine harvesting/CH S	ha	11.1	11.1	10.8	11.7	6	9.9	10.2	8.7
concentrate feed IP, 16% crude protein, for dairy cows, at consumer/kg/CH U	kg	18360	17340	18360	18360	18870	18870	18360	18360
Cubicle housing, cattle, wood construction, non-insulated, at farm/p/CH/I U	p	0.19159	0.18095	0.19159	0.19159	0.19691	0.19691	0.19159	0.19159
Cyclic N-compounds, at regional storehouse/CH S	kg	2.8399	2.8402	2.7646	2.9903	0.16627	2.5383	2.6154	2.2368
Diammonium phosphate, as N, at regional storehouse/RER S	kg	50.173	54.304	48.933	48.993	48.779	47.024	50.053	53.684

Table A3.7 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Diammonium phosphate, as P2O5, at regional storehouse/RER S	kg	39.201	33.773	58.359	56.594	37.928	44.468	57.098	51.593
Dicamba, at regional storehouse/CH S	kg	0.10067	0.10067	0.097946	0.10611		0.089784	0.092505	0.078901
Diesel combustion, in tractor/kg/CH U	kg	4358	4271.2	4282.2	4382.5	3792.5	4266	4266.5	4148
Diesel, at regional storage/CH S	kg	4358	4271.2	4282.2	4382.5	3792.5	4266	4266.5	4148
Dinitroaniline-compounds, at regional storehouse/CH S	kg	3.9604	3.9684	3.7953	3.9304	2.949	3.7658	3.7394	3.9262
Diphenylether-compounds, at regional storehouse/CH S	kg	0.10949	0.10949	0.10653	0.1154	0.25518	0.09765	0.10061	0.085813
Dried-forage storage, wood, non-insulated, without ventilation, at farm/m3/CH/I U	m3	4.1869	4.1776	4.1662	4.042	5.037	4.3131	4.2978	4.7122
Dried-forage storage,wood,non-insulated,with cool air ventilation,at farm/m3/CH/I U	m3	8.4526	8.4216	9.1687	8.7551	11.285	8.8731	9.6073	7.8469
Dung slab/CH/I S	m2	2.1061	1.9891	2.1061	2.1061	2.1646	2.1646	2.1061	2.1061
Electricity, low voltage, at grid/CH S	kWh	15974	16169	16216	16066	16863	15984	16376	15811
Exercise yard, floored, perforated, at farm/m2/CH/I U	m2	3.7152	3.5088	3.7152	3.7152	3.8184	3.8184	3.7152	3.7152
Feed-concentrate silo, steel, at farm/m3/CH/I U	m3	1.8193	1.8193	1.8193	1.8193	1.8193	1.8193	1.8193	1.8193
Garage, wood, non-insulated, fire-protected, at farm/m2/CH/I U	m2	1.2407	1.2262	1.2179	1.2431	1.1208	1.217	1.2264	1.1889
Glyphosate, at regional storehouse/CH S	kg	0.26256	0.26438	0.26413	0.25836	0.30395	0.26696	0.27577	0.27123
Grabcrane, at farm/p/CH/I U	p	0.016973	0.016911	0.018411	0.017581	0.02266	0.017817	0.019292	0.015757
Grain drying, low temperature/CH S	kg	863.4	863.4	840.06	910.07	1342.5	770.06	793.39	676.72
Grass seed IP, at regional storehouse/CH S	kg	1.9458	1.9955	1.8451	1.9565	1.6901	1.8962	1.9445	1.7819
Harvester, production/CH/I S	kg	13.508	13.91	13.011	13.457	12.282	13.107	13.814	12.324
Horizontal silo, concrete, at farm/m3/CH/I U	m3	7.0295	7.0171	6.7933	6.6282	8.1604	7.1974	6.9685	8.3539
Irrigation pipeline, at farm/m/CH/I U	m	11.375	11.375	60.125	60.125	11.375	11.375	9.75	
Light fuel oil, burned in boiler 10kW, non-modulating/CH S	MJ	16999	16999	16999	16999	16999	16999	16999	16999
Linuron, at regional storehouse/CH S	kg	0.21645	0.21645	0.2106	0.22815		0.19305	0.1989	0.16965
Magnesium oxide, at plant/RER S	kg	424.66	385.54	509.42	497.75	456.54	447.94	452.61	454.22
Maize seed IP, at regional storehouse/CH S	kg	0.567	0.567	0.486	0.486	0.567	0.567	0.486	0.81
MCPA, at regional storehouse/CH S	kg	1.7264	1.7275	1.6849	1.8087	0.18111	1.5593	1.607	1.3921
Milk cooling tank, at farm/m3/CH/I U	m3	0.08088	0.076387	0.08088	0.08088	0.083127	0.083127	0.08088	0.08088
Milking parlour, at farm/p/CH/I U	p	0.031932	0.030158	0.031932	0.031932	0.032819	0.032819	0.031932	0.031932
Monoammonium phosphate, as N, at regional storehouse/RER S	kg	50.173	54.304	48.933	48.993	48.779	47.024	50.053	53.684
Monoammonium phosphate, as P2O5, at regional storehouse/RER S	kg	39.201	33.773	58.359	56.594	37.928	44.468	57.098	51.593
Nitrile-compounds, at regional storehouse/CH S	kg	1.9024	1.9031	1.8491	1.9897	0.23753	1.7138	1.7584	1.5441
Organophosphorus-compounds, at regional storehouse/CH S	kg					1.3655			

Table A3.7 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Passenger car/RER/I S	p	0.058111	0.056655	0.057931	0.058044	0.058486	0.058247	0.057906	0.058363
Pesticide unspecified, at regional storehouse/CH S	kg	0.52023	0.52023	0.49704	0.53284	0.2862	0.4725	0.47318	0.45852
Petrol combustion, unleaded, in motor mower/kg/CH U	kg	389.73	389.99	386.08	388.88	382.26	382.98	388.84	387.79
Petrol, unleaded, at regional storage/CH S	kg	389.73	389.99	386.08	388.88	382.26	382.98	388.84	387.79
Phenoxy-compounds, at regional storehouse/CH S	kg	2.3702	2.3745	2.3184	2.4421	0.77565	2.1977	2.2541	2.054
Phosphate rock, as P2O5, beneficiated, dry, at plant/MA S	kg	67.202	57.897	100.04	97.019	65.02	76.231	97.882	88.445
Pipeline milking system, at farm/p/CH/I U	p	1.1009	1.0397	1.1009	1.1009	1.1314	1.1314	1.1009	1.1009
Potassium chloride, as K2O, at regional storehouse/RER S	kg	243.43	188.07	527.84	494.28	341.95	353.37	559.92	470.19
Potassium sulphate, as K2O, at regional storehouse/RER S	kg	15.538	12.004	33.692	31.55	21.827	22.555	35.739	30.012
Pumping station, concrete walls, non-insulated, at farm/m3/CH/I U	m3	0.000875	0.000875	0.004625	0.004625	0.000875	0.000875	0.00075	
Pyretroid-compounds, at regional storehouse/CH S	kg					0.075041			
Rape seed IP, at regional storehouse/CH S	kg					0.33			
Shed, large, wood, non-insulated, fire-unprotected, at farm/m2/CH/I U	m2	4.1874	4.1514	4.0908	4.2259	3.0713	4.0312	4.0763	3.8834
Single superphosphate, as P2O5, at regional storehouse/RER S	kg	5.6001	4.8247	8.3369	8.0849	5.4183	6.3525	8.1568	7.3704
Slurry store and processing/CH/I S	m3	8.6056	8.1275	8.6056	8.6056	8.8446	8.8446	8.6056	8.6056
Slurry tanker, production/CH/I S	kg	62.046	58.599	62.046	62.046	63.77	63.77	62.046	62.046
Sowing/CH S	ha	2.1	2.1	1.8	1.8	2.1	2.1	1.8	3
Soya bean meal, at oil mill/kg/CH U	kg	9000	8500	9000	9000	9250	9250	9000	9000
Straw IP, at farm/CH S	kg	7080	7080	7080	7080	7080	7080	7080	7080
Tap water, at user/CH S	kg	2947.3	2832.9	2997	2997	3079.1	3058.7	2997	2997
Tethered housing, cattle, concrete wall, at farm/p/CH/I U	p	0.52841	0.49905	0.52841	0.52841	0.54309	0.54309	0.52841	0.52841
Thomas meal, as P2O5, at regional storehouse/RER S	kg	14	12.062	20.842	20.212	13.546	15.881	20.392	18.426
Tower silo, plastic, at farm/m3/CH/I U	m3	7.0295	7.0171	6.7933	6.6282	8.1604	7.1974	6.9685	8.3539
Tractor, production/CH/I S	kg	511.22	499.75	503.13	512.81	459.52	503.7	502.49	492.06
Trailer, production/CH/I S	kg	297.15	283.47	296.34	298.5	312.23	300.84	294.98	292.02
Transport, barge/RER S	tkm	9385.3	8759.9	9298.9	9607.5	6432.3	8408.8	8624.4	8405.8
Transport, freight, rail/CH S	tkm	1166.8	1079	1229.4	1255.4	857.11	1083.2	1151.7	1108.5
Transport, lorry 16-32t, EURO3/RER S	tkm	1796.8	1674	1859.4	1885.4	1504.6	1730.7	1781.7	1738.5
Transport, tractor and trailer/CH S	tkm	247.8	247.8	247.8	247.8	247.8	247.8	247.8	247.8
Transport, van <3.5t/CH S	tkm	1.1259	1.1273	1.0889	1.1638	0.42532	1.0263	1.0442	0.95871
Triazine-compounds, at regional storehouse/CH S	kg	0.13833	0.13836	0.12245	0.12475	0.10959	0.13522	0.12101	0.17745
Triple superphosphate, as P2O5, at regional storehouse/RER S	kg	114.8	98.907	170.91	165.74	111.08	130.23	167.22	151.09
Unloading sucker blower, telescopic handler, at farm/p/CH/I U	p	0.011315	0.011274	0.012274	0.01172	0.015107	0.011878	0.012861	0.010505

Table A3.7 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Urea, as N, at regional storehouse/RER S	kg	590.88	551.74	574.1	595.59	395.62	523.54	529.52	517.76
Wheat seed IP, at regional storehouse/CH S	kg	19.98	19.98	19.44	21.06		17.82	18.36	15.66
Emissions to air (low population)									
Ammonia	kg	797.83	747.13	788.43	799.34	697.63	784.44	773.63	768.92
Carbon dioxide, fossil	kg	927.68	866.23	901.34	935.07	621.12	821.96	831.35	812.88
Dinitrogen monoxide	kg	132.99	125.6	130.79	133.07	116.69	127.64	126.62	110.87
Methane, biogenic	kg	337.92	337.82	337.92	337.92	337.97	337.97	337.92	337.92
Nitrogen oxides	kg	27.928	26.375	27.466	27.946	24.504	26.805	26.59	23.283
Emissions to water									
Cadmium, ion, groundwater	kg	0.000721	0.0006653	0.000864	0.000855	0.000702	0.000759	0.000855	0.00082
Cadmium, ion, river	kg	3.9E-14	3.606E-14	4.52E-14	4.69E-14	2.35E-14	3.89E-14	3.45E-14	3.44E-14
Chromium, ion, groundwater	kg	0.50231	0.48646	0.53295	0.53232	0.49538	0.5058	0.52958	0.52371
Chromium, ion, river	kg	4.98E-12	4.83E-12	5.11E-12	5.35E-12	3.04E-12	4.75E-12	3.91E-12	4.03E-12
Copper, ion, groundwater	kg	0.064234	0.061606	0.068214	0.068413	0.058792	0.063497	0.067205	0.065785
Copper, ion, river	kg	2.86E-12	2.746E-12	2.94E-12	3.09E-12	1.62E-12	2.68E-12	2.23E-12	2.27E-12
Lead, groundwater	kg	0.000932	0.0008479	0.001102	0.001103	0.000772	0.000929	0.001061	0.000997
Lead, river	kg	3.35E-13	3.049E-13	3.83E-13	4.01E-13	1.72E-13	3.16E-13	2.84E-13	2.78E-13
Mercury, groundwater	kg	1.7E-08	1.715E-08	1.65E-08	1.92E-08	8.65E-09	1.63E-08	1.66E-08	1.54E-08
Mercury, river	kg	1.01E-17	1.018E-17	9.46E-18	1.15E-17	3.18E-18	9.17E-18	7.33E-18	7.11E-18
Nickel, ion, river	kg	2.52E-12	2.399E-12	2.54E-12	2.69E-12	1.3E-12	2.29E-12	1.89E-12	1.92E-12
Nitrate, groundwater	kg	4527.3	4353.9	4347.7	4335.8	4573.3	4414.5	4374.3	519.53
Phosphate, groundwater	kg	19.785	19.879	18.438	18.18	20.728	19.578	19.283	
Phosphorus, river	kg	0.000199	0.000194	0.000169	0.000166	0.000181	0.000197	0.000129	
Zinc, ion, groundwater	kg	0.22658	0.21169	0.24719	0.2487	0.19217	0.22196	0.23921	0.23061
Zinc, ion, river	kg	3.89E-12	3.634E-12	4.1E-12	4.32E-12	2.04E-12	3.6E-12	3.05E-12	3.07E-12
Emissions to soil (agricultural)									
2,4-D	kg	0.96305	0.96394	0.93335	0.99268	0.21675	0.88233	0.8976	0.8308
Acetamiprid	kg					0.011379			
Acibenzolar-s-methyl	kg	0.030273	0.030273	0.029454	0.031909		0.027	0.027818	0.023727
Alachlor	kg	0.1764	0.1764	0.1512	0.1512	0.1764	0.1764	0.1512	0.252
Alpha-cypermethrin	kg					0.008152			
Amidosulfuron	kg	0.023844	0.023897	0.023452	0.024596	0.008865	0.022222	0.022917	0.020597
Asulam	kg	0.70016	0.70502	0.70434	0.68895	0.81054	0.7119	0.73539	0.72328
Atrazine	kg	0.27363	0.27363	0.23454	0.23454	0.27363	0.27363	0.23454	0.3909
Azoxystrobin	kg	0.35318	0.35318	0.34363	0.37227		0.315	0.32454	0.27682
Bentazone	kg	2.3865	2.3865	2.2902	2.4614	0.27468	2.1582	2.176	2.0476
Bifenthrin	kg					0.006621			
Bromoxynil	kg	0.37104	0.37104	0.35591	0.38242	0.0441	0.3357	0.33824	0.31925
Cadmium	kg	0.012542	0.010408	0.020496	0.019291	0.002822	0.015119	0.020253	0.018712
Carbetamide	kg					0.54545			
Carfentrazone-ethyl	kg	0.025227	0.025227	0.024545	0.026591		0.0225	0.023182	0.019773
Chlorothalonil	kg	3.9733	3.9733	3.8659	4.188		3.5437	3.6511	3.1142
Chlorotoluron	kg	0.94854	0.94854	0.92291	0.99982		0.846	0.87163	0.74345
Chlorpyrifos	kg					0.49655			
Clethodim	kg					0.065454			
Clomazone	kg					0.10424			
Clopyralid	kg	0.008752	0.0088128	0.008804	0.008612	0.010132	0.008899	0.009192	0.009041
Cycloxydim	kg	0.01575	0.01575	0.0135	0.0135	0.097568	0.01575	0.0135	0.0225
Cypermethrin	kg					0.045517			
Cyproconazole	kg	0.1614	0.1614	0.15704	0.17013		0.14395	0.14832	0.12651
Cyprodinil	kg	0.60545	0.60545	0.58909	0.63818		0.54	0.55636	0.47454
Deltamethrin	kg					0.012165			
Diazinon	kg					0.28965			
Dicamba	kg	0.10067	0.10067	0.097946	0.10611		0.089784	0.092505	0.078901
Dichlobenil	kg	0.098461	0.099144	0.099048	0.096884	0.11398	0.10011	0.10341	0.10171

Table A3.7 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Difenoconazole	kg	0.37841	0.37841	0.36818	0.39886		0.3375	0.34773	0.29659
Diflufenican	kg	0.10949	0.10949	0.10653	0.1154		0.09765	0.10061	0.085813
Dimefuron	kg					0.27273			
Dimethenamid	kg	0.26092	0.26092	0.22365	0.22365	0.26092	0.26092	0.22365	0.37275
Epoxiconazole	kg	0.88609	0.88609	0.86214	0.93399		0.7903	0.81424	0.6945
Famoxadone	kg	0.30273	0.30273	0.29454	0.31909		0.27	0.27818	0.23727
Fenpropimorph	kg	0.70636	0.70636	0.68727	0.74454		0.63	0.64909	0.55363
Fluazifop	kg					0.15341			
Fluroxypyr	kg	0.19135	0.19135	0.18618	0.20169		0.17066	0.17584	0.14998
Flusilazole	kg	0.32331	0.32331	0.31457	0.34079		0.28836	0.2971	0.25341
Glyphosate	kg	0.26256	0.26438	0.26413	0.25836	0.30395	0.26696	0.27577	0.27123
Haloxyfop-P-methyl	kg					0.029454			
Hexaconazole	kg	0.1892	0.1892	0.18409	0.19943		0.16875	0.17386	0.14829
Ioxynil	kg	0.17155	0.17155	0.16691	0.18082		0.153	0.15764	0.13445
Isoproturon	kg	2.1595	2.1595	2.1011	2.2762		1.926	1.9844	1.6925
Kresoxim-methyl	kg	0.25227	0.25227	0.24545	0.26591		0.225	0.23182	0.19773
Lambda-cyhalothrin	kg					0.006724			
Lead	kg	0.026555	0.024038	0.031837	0.031691		0.026624	0.030656	0.028931
Linuron	kg	0.21645	0.21645	0.2106	0.22815		0.19305	0.1989	0.16965
MCPA	kg	1.7264	1.7275	1.6849	1.8087		1.5593	1.607	1.3921
MCPB	kg	0.26256	0.26438	0.26413	0.25836		0.26696	0.27577	0.27123
Mecoprop-P	kg	0.95325	0.95478	0.93476	0.98935		0.8777	0.9049	0.80203
Metazachlor	kg					1.3003			
Metolachlor, (S)	kg	0.16632	0.16632	0.14256	0.14256	0.16632	0.16632	0.14256	0.2376
Metoxuron	kg	0.48436	0.48436	0.47127	0.51054		0.432	0.44509	0.37964
Metsulfuron-methyl	kg	0.004488	0.0044884	0.004367	0.004731		0.004003	0.004125	0.003518
Napropamide	kg					1.6028			
Nickel	kg	0.075659	0.06858	0.084077	0.084517		0.070841	0.078409	0.076133
Nicosulfuron	kg	0.00525	0.00525	0.0045	0.0045		0.00525	0.0045	0.0075
Pendimethalin	kg	3.476	3.484	3.324	3.4198		3.3338	3.2944	3.5465
Phosalone	kg					0.57931			
Pirimicarb	kg					0.31034			
Pronamide	kg					0.17045			
Propaquizafop	kg					0.047727			
Propiconazole	kg	0.12614	0.12614	0.12273	0.13295		0.1125	0.11591	0.098863
Pyraclostrobin (prop)	kg	0.46973	0.46973	0.45703	0.49512		0.41895	0.43164	0.36817
Quizalofop ethyl ester	kg					0.020454			
Simazine	kg	0.105	0.105	0.09	0.09	0.105	0.105	0.09	0.15
Spinosad	kg					0.039724			
Sulcotrione	kg	0.063	0.063	0.054	0.054	0.063	0.063	0.054	0.09
Tepraloxydim	kg					0.020454			
Thiacloprid	kg					0.079448			
Thifensulfuron-methyl	kg	0.028845	0.028867	0.028082	0.030019	0.004587	0.026221	0.026888	0.023934
Triclopyr	kg	0.035008	0.035251	0.035217	0.034447	0.040527	0.035595	0.036769	0.036164
Trifloxystrobin	kg	0.1892	0.1892	0.18409	0.19943		0.16875	0.17386	0.14829
Trifluralin	kg	0.48436	0.48436	0.47127	0.51054	0.78545	0.432	0.44509	0.37964

Table A3.8: *inventories of all mixed farm scenarios for the Greifensee region, according to ecoinvent and SALCA nomenclature (continued pages 167 to 170).*

		Scenario					
Flow	Unit	Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Resources							
Water, river	l			2205000			
Materials/fuels							
[sulfonyl]urea-compounds, at regional storehouse/CH S	kg	2.6463	2.6462	2.0591	2.2578	0.2077	0.77359
[thio]carbamate-compounds, at regional storehouse/CH S	kg	2.0024	1.9958	1.8479	1.0869	1.3803	2.6884
2,4-D, at regional storehouse/CH S	kg	0.75526	0.75404	0.61104	0.70013	0.25038	0.30263
Acetamide-anillide-compounds, at regional storehouse/CH S	kg	1.0763	1.0763	2.0115	0.68988	3.1001	4.185
Agricultural machinery, general, production/CH/I S	kg	408.63	406.99	355.83	405.92	242.11	295.66
Agricultural machinery, tillage, production/CH/I S	kg	140.43	140.43	128.92	99.33	84.577	151.85
Ammonium nitrate, as N, at regional storehouse/RER S	kg	1641.8	1484	1396.4	1537.4	1318.9	1581.8
Ammonium sulphate, as N, at regional storehouse/RER S	kg	126.3	114.15	107.41	118.26	101.46	121.68
Atrazine, at regional storehouse/CH S	kg	0.31272	0.31272	0.31272	0.31272	0.42999	0.31272
Baling/CH S	p	0.096677	0.096677	0.096677	0.064451		0.064451
Benzimidazole-compounds, at regional storehouse/CH S	kg	3.1449	3.1449	2.3039	2.4699		0.80472
Benzo[thia]diazole-compounds, at regional storehouse/CH S	kg	1.8771	1.8771	1.4718	1.6455	0.43164	0.60339
Bucket milking system, at farm/p/CH/I U	p	1.1009	1.0703	1.1314	1.1314	1.162	1.1009
Calcium ammonium nitrate, as N, at regional storehouse/RER S	kg	820.92	741.98	698.18	768.68	659.47	790.92
Cattle for breeding, Swiss integrated production, at farm gate/CH U	kg	5940	5775	6105	6105	6270	5940
Combine harvesting/CH S	ha	8.1	8.1	8.1	6.9	4.2	7.5
concentrate feed IP, 16% crude protein, for dairy cows, at consumer/kg/CH U	kg	18360	17850	18870	18870	19380	18360
Cubicle housing, cattle, wood construction, non-insulated, at farm/p/CH/I U	p	0.19159	0.18627	0.19691	0.19691	0.20224	0.19159
Cyclic N-compounds, at regional storehouse/CH S	kg	2.5601	2.5597	1.8975	1.7878	0.12976	1.0578
Diammonium phosphate, as N, at regional storehouse/RER S	kg	50.928	52.037	47.222	51.294	51.523	49.493
Diammonium phosphate, as P2O5, at regional storehouse/RER S	kg	70.705	67.757	73.336	74.546	48.442	84.796
Dicamba, at regional storehouse/CH S	kg	0.07346	0.07346	0.054414	0.062577		0.013604
Diesel combustion, in tractor/kg/CH U	kg	4226.3	4168.8	4096.2	4022.7	3638	4137.1
Diesel, at regional storage/CH S	kg	4226.3	4168.8	4096.2	4022.7	3638	4137.1
Dinitroaniline-compounds, at regional storehouse/CH S	kg	3.451	3.44	3.3587	3.4862	3.1391	3.0883
Diphenylether-compounds, at regional storehouse/CH S	kg	0.16779	0.16779	0.20443	0.068059	0.17863	0.36586
Dried-forage storage, wood, non-insulated, without ventilation, at farm/m3/CH/I U	m3	4.1817	4.1348	4.3169	4.9372	5.5392	4.2547
Dried-forage storage,wood,non-insulated,with cool air ventilation,at farm/m3/CH/I U	m3	7.65	7.4936	8.1005	10.167	9.8169	7.8929
Dung slab/CH/I S	m2	2.1061	2.0476	2.1646	2.1646	2.2231	2.1061
Electricity, low voltage, at grid/CH S	kWh	15701	15761	15722	16475	16234	15789
Exercise yard, floored, perforated, at farm/m2/CH/I U	m2	3.7152	3.612	3.8184	3.8184	3.9216	3.7152
Feed-concentrate silo, steel, at farm/m3/CH/I U	m3	1.8193	1.8193	1.8193	1.8193	1.6265	1.8193
Garage, wood, non-insulated, fire-protected, at farm/m2/CH/I U	m2	1.2042	1.1904	1.1584	1.1746	1.0603	1.1978

Table A3.8 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Glyphosate, at regional storehouse/CH S	kg	0.24134	0.23884	0.24296	0.2981	0.29297	0.24777
Grabcrane, at farm/p/CH/I U	p	0.015362	0.015047	0.016266	0.020416	0.019713	0.015849
Grain drying, low temperature/CH S	kg	630.05	630.05	936.58	536.71	939.75	1459.2
Grass seed IP, at regional storehouse/CH S	kg	2.1606	2.1606	1.8573	1.8159	1.4792	2.2612
Harvester, production/CH/I S	kg	11.444	11.444	10.259	13.082	9.586	11.94
Harvesting, by complete harvester, beets/CH S	ha	3.3	3.3	2.1			3.6
Horizontal silo, concrete, at farm/m3/CH/I U	m3	7.2311	7.1686	7.411	8.2361	9.6625	7.3281
Irrigation pipeline, at farm/m/CH/I U				11.375			
Light fuel oil, burned in boiler 10kW, non-modulating/CH S	MJ	16999	16999	16999	16999	14068	16999
Linuron, at regional storehouse/CH S	kg	0.15795	0.15795	0.117	0.13455		0.02925
Magnesium oxide, at plant/RER S	kg	655.38	612.04	659.69	597.35	605.31	691.77
Maize seed IP, at regional storehouse/CH S	kg	0.648	0.648	0.648	0.648	0.891	0.648
MCPA, at regional storehouse/CH S	kg	1.2894	1.2879	0.99338	1.1535	0.17456	0.35978
Milk cooling tank, at farm/m3/CH/I U	m3	0.08088	0.078633	0.083127	0.083127	0.085373	0.08088
Milking parlour, at farm/p/CH/I U	p	0.031932	0.031045	0.032819	0.032819	0.033706	0.031932
Monoammonium phosphate, as N, at regional storehouse/RER S	kg	50.928	52.037	47.222	51.294	51.523	49.493
Monoammonium phosphate, as P2O5, at regional storehouse/RER S	kg	70.705	67.757	73.336	74.546	48.442	84.796
Nitrile-compounds, at regional storehouse/CH S	kg	1.4251	1.4242	1.1206	1.2561	0.23478	0.46058
Organophosphorus-compounds, at regional storehouse/CH S	kg	0.32708	0.32708	0.68607		0.95586	1.7223
Passenger car/RER/I S	p	0.060663	0.059714	0.060231	0.058584	0.059746	0.060866
Pesticide unspecified, at regional storehouse/CH S	kg	0.48146	0.48146	0.44534	0.36443	0.26897	0.43271
Petrol combustion, unleaded, in motor mower/kg/CH U	kg	407.31	404.5	391.98	386.23	378.05	411.19
Petrol, unleaded, at regional storage/CH S	kg	407.31	404.5	391.98	386.23	378.05	411.19
Phenoxy-compounds, at regional storehouse/CH S	kg	1.8736	1.8678	1.5574	1.8229	0.7891	0.88314
Phosphate rock, as P2O5, beneficiated, dry, at plant/MA S	kg	121.21	116.15	125.72	127.79	83.044	145.36
Pipeline milking system, at farm/p/CH/I U	p	1.1009	1.0703	1.1314	1.1314	1.162	1.1009
Potassium chloride, as K2O, at regional storehouse/RER S	kg	647.41	636.49	709	782.86	426.76	794.45
Potassium sulphate, as K2O, at regional storehouse/RER S	kg	41.324	40.627	45.255	49.97	27.24	50.71
Pumping station, concrete walls, non-insulated, at farm/m3/CH/I U				0.000875			
Pyretroid-compounds, at regional storehouse/CH S	kg	0.020586	0.020586	0.039365		0.052529	0.097499
Pyridazine-compounds, at regional storehouse/CH S	kg	0.67414	0.67414	0.429			0.73543
Rape seed IP, at regional storehouse/CH S				0.1155		0.231	0.33
Shed, large, wood, non-insulated, fire-unprotected, at farm/m2/CH/I U	m2	3.8317	3.7954	3.5614	3.7022	2.8869	3.3736
Single superphosphate, as P2O5, at regional storehouse/RER S	kg	10.101	9.6796	10.477	10.649	6.9203	12.114
Slurry store and processing/CH/I S	m3	8.6056	8.3665	8.8446	8.8446	9.0836	8.6056
Slurry tanker, production/CH/I S	kg	62.046	60.323	63.77	63.77	65.493	62.046
Sowing/CH S	ha	5.7	5.7	4.5	2.4	3.3	6
Soya bean meal, at oil mill/kg/CH U	kg	9000	8750	9250	9250	9500	9000
Straw IP, at farm/CH S	kg	7080	7080	7080	7080	13680	7080
Sugar beet seed IP, at regional storehouse/CH S	kg	0.0693	0.0693	0.0441			0.0756
Tap water, at user/CH S	kg	2962.2	2924.9	3089.6	3089.6	3172	2992.1
Tethered housing, cattle, concrete wall, at farm/p/CH/I U	p	0.52841	0.51373	0.54309	0.54309	0.55776	0.52841
Thomas meal, as P2O5, at regional storehouse/RER S	kg	25.252	24.199	26.192	26.623	17.301	30.284
Tower silo, plastic, at farm/m3/CH/I U	m3	7.2311	7.1686	7.411	8.2361	9.6625	7.3281

Table A3.8 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Tractor, production/CH/I S	kg	499.82	492.42	487.28	481.92	445.66	492.68
Trailer, production/CH/I S	kg	292.57	285.55	305.46	294.38	309.44	311.33
Transport, barge/RER S	tkm	9316.1	8470.7	8045.6	8817.6	7444.8	9112.2
Transport, freight, rail/CH S	tkm	1278.7	1175.3	1150.2	1241.3	1010.2	1296.2
Transport, lorry 16-32t, EURO3/RER S	tkm	1908.7	1787.8	1797.7	1888.8	1675.2	1926.2
Transport, tractor and trailer/CH S	tkm	2623.8	2623.8	1759.8	247.8	478.8	2839.8
Transport, van <3.5t/CH S	tkm	1.0098	1.0089	0.85969	0.79594	0.3971	0.69048
Triazine-compounds, at regional storehouse/CH S	kg	1.0694	1.0693	0.72781	0.14282	0.1699	1.1361
Triple superphosphate, as P2O5, at regional storehouse/RER S	kg	207.07	198.43	214.77	218.31	141.87	248.33
Unloading sucker blower, telescopic handler, at farm/p/CH/I U	p	0.010241	0.010032	0.010844	0.01361	0.013142	0.010566
Urea, as N, at regional storehouse/RER S	kg	568.33	513.68	483.35	532.16	456.56	547.56
Wheat seed IP, at regional storehouse/CH S	kg	14.58	14.58	10.8	12.42		2.7
Emissions to air (low population)							
Ammonia	kg	792.11	761	770.69	776.94	752.98	785.45
Carbon dioxide, fossil	kg	892.28	806.48	758.86	835.5	716.8	859.67
Dinitrogen monoxide	kg	139.47	132.44	130.12	127.58	123.46	140.42
Methane, biogenic	kg	337.92	337.87	337.97	337.97	338.02	337.92
Nitrogen oxides	kg	29.288	27.812	27.325	26.793	25.927	29.487
Emissions to water							
Cadmium, ion, groundwater	kg	0.000933	0.0009129	0.00094	0.000946	0.000794	0.000994
Cadmium, ion, river	kg	5.15E-14	5.076E-14	4.12E-14	3.46E-14	3.11E-14	4.66E-14
Chromium, ion, groundwater	kg	0.54989	0.54225	0.54805	0.54802	0.52201	0.56047
Chromium, ion, river	kg	5.57E-12	5.522E-12	4.4E-12	3.67E-12	3.74E-12	4.82E-12
Copper, ion, groundwater	kg	0.070514	0.06858	0.069175	0.07053	0.063498	0.072419
Copper, ion, river	kg	3.2E-12	3.136E-12	2.49E-12	2.12E-12	2.04E-12	2.79E-12
Lead, groundwater	kg	0.001204	0.0011334	0.001158	0.001225	0.000921	0.001301
Lead, river	kg	4.41E-13	4.18E-13	3.37E-13	2.97E-13	2.39E-13	4.05E-13
Mercury, groundwater	kg	1.67E-08	1.669E-08	1.5E-08	1.29E-08	5.76E-09	1.33E-08
Mercury, river	kg	1.01E-17	1.016E-17	7.21E-18	5.18E-18	2.47E-18	6.85E-18
Nickel, ion, river	kg	2.77E-12	2.661E-12	2.09E-12	1.8E-12	1.69E-12	2.4E-12
Nitrate, groundwater	kg	4834.6	4687.5	4597.5	4465.2	4719	4892.9
Phosphate, groundwater	kg	22.373	22.264	21.455	21.941	22.507	23.32
Phosphorus, river	kg	0.000283	0.0002807	0.000284	0.0002	0.000341	0.000294
Zinc, ion, groundwater	kg	0.26012	0.24725	0.24999	0.26026	0.2172	0.27179
Zinc, ion, river	kg	4.55E-12	4.354E-12	3.47E-12	3.01E-12	2.69E-12	4.04E-12
Emissions to soil (agricultural)							
2,4-D	kg	0.75526	0.75404	0.61104	0.70013	0.25038	0.30263
Acetamiprid	kg			0.003983		0.007966	0.011379
Acibenzolar-s-methyl	kg	0.022091	0.022091	0.016364	0.018818		0.004091
Alachlor	kg	0.2016	0.2016	0.2016	0.2016	0.2772	0.2016
Alpha-cypermethrin	kg	0.002167	0.002167	0.004232		0.005706	0.010516
Amidosulfuron	kg	0.01885	0.018777	0.015835	0.018756	0.008545	0.009414
Asulam	kg	0.64356	0.6369	0.64789	0.79493	0.78126	0.66072
Atrazine	kg	0.31272	0.31272	0.31272	0.31272	0.42999	0.31272
Azoxystrobin	kg	0.25773	0.25773	0.19091	0.21954		0.047727
Bentazone	kg	1.855	1.855	1.4554	1.6267	0.43164	0.5993
Bifenthrin	kg	0.00176	0.00176	0.003437		0.004635	0.008541
Bromoxynil	kg	0.28898	0.28898	0.22713	0.25364	0.0693	0.094582
Cadmium	kg	0.013022	0.011847	0.013888	0.029759	0.010772	0.001461
Carbetamide	kg			0.19091		0.38182	0.54545

Table A3.8 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Carbofuran	kg	0.11	0.11	0.07			0.12
Carfentrazone-ethyl	kg	0.018409	0.018409	0.013636	0.015682		0.003409
Chloridazon	kg	0.67414	0.67414	0.429			0.73543
Chlorothalonil	kg	2.8994	2.8994	2.1477	2.4699		0.53693
Chlorotoluron	kg	0.69218	0.69218	0.51273	0.58964		0.12818
Chlorpyrifos	kg	0.132	0.132	0.25779		0.34759	0.64055
Chromium	kg	0.004012		0.058035	0.11587		0.10909
Clethodim	kg	0.033943	0.033943	0.044509		0.045818	0.10248
Clomazone	kg			0.036483		0.072965	0.10424
Clopyralid	kg	0.028787	0.028704	0.021299	0.009937	0.009766	0.030887
Cycloxydim	kg	0.046286	0.046286	0.064636	0.018	0.082022	0.13067
Cypermethrin	kg	0.0121	0.0121	0.023631		0.031862	0.058717
Cyproconazole	kg	0.11778	0.11778	0.087245	0.10033		0.021811
Cyprodinil	kg	0.44182	0.44182	0.32727	0.37636		0.081818
Deltamethrin	kg	0.00401	0.0040095	0.006809		0.008516	0.016539
Desmedipham	kg	0.046341	0.046341	0.02949			0.050554
Diazinon	kg	0.077	0.077	0.15038		0.20276	0.37365
Dicamba	kg	0.07346	0.07346	0.054414	0.062577		0.013604
Dichlobenil	kg	0.090501	0.089564	0.09111	0.11179	0.10986	0.092913
Difenoconazole	kg	0.27614	0.27614	0.20454	0.23523		0.051136
Diflufenican	kg	0.079895	0.079895	0.059182	0.068059		0.014795
Dimefuron	kg			0.095454		0.19091	0.27273
Dimethenamid	kg	0.4764	0.4764	0.4116	0.2982	0.41002	0.4926
Dimethoate	kg	0.044	0.044	0.028			0.048
Epoxiconazole	kg	0.64661	0.64661	0.47897	0.55081		0.11974
Ethofumesate	kg	0.24547	0.24547	0.15621			0.26779
Famoxadone	kg	0.22091	0.22091	0.16364	0.18818		0.040909
Fenpropimorph	kg	0.51545	0.51545	0.38182	0.43909		0.095454
Fluazifop	kg	0.053036	0.053036	0.087443		0.10739	0.21127
Fluroxypyr	kg	0.13963	0.13963	0.10343	0.11895		0.025858
Flusilazole	kg	0.23593	0.23593	0.17476	0.20098		0.043691
Glyphosate	kg	0.24134	0.23884	0.24296	0.2981	0.29297	0.24777
Haloxyfop-P-methyl	kg	0.010183	0.010183	0.016789		0.020618	0.040563
Heptenophos	kg	0.035585	0.035585	0.022645			0.03882
Hexaconazole	kg	0.13807	0.13807	0.10227	0.11761		0.025568
Ioxynil	kg	0.12518	0.12518	0.092727	0.10664		0.023182
Isoproturon	kg	1.5758	1.5758	1.1673	1.3424		0.29182
Kresoxim-methyl	kg	0.18409	0.18409	0.13636	0.15682		0.034091
Lambda-cyhalothrin	kg	0.00165	0.00165	0.003403		0.004707	0.008524
Lead	kg	0.031076	0.029051	0.029488	0.036449	0.024477	0.027742
Lenacil	kg	0.45257	0.45257	0.288			0.49371
Linuron	kg	0.15795	0.15795	0.117	0.13455		0.02925
MCPA	kg	1.2894	1.2879	0.99338	1.1535	0.17456	0.35978
MCPB	kg	0.24134	0.23884	0.24296	0.2981	0.29297	0.24777
Mecoprop-P	kg	0.73734	0.73524	0.60002	0.7057	0.24574	0.30688
Metamitron	kg	0.92409	0.92409	0.58806			1.0081
Metazachlor	kg			0.45512		0.91023	1.3003
Methidathion	kg	0.0385	0.0385	0.0245			0.042
Methomyl	kg	0.055	0.055	0.035			0.06
Metolachlor, (S)	kg	0.39826	0.39826	0.32256	0.19008	0.26136	0.41719
Metoxuron	kg	0.35345	0.35345	0.26182	0.30109		0.065454
Metsulfuron-methyl	kg	0.003275	0.0032753	0.002426	0.00279		0.000607

Table 3.8 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Napropamide	kg			0.56098		1.122	1.6028
Nickel	kg	0.065454	0.05784	0.061861	0.092669	0.065785	0.053956
Nicosulfuron	kg	0.006	0.006	0.006	0.006	0.00825	0.006
Pendimethalin	kg	3.0975	3.0866	2.822	3.1851	2.5893	2.2373
Phenmedipham	kg	0.72225	0.72225	0.45961			0.7879
Phosalone	kg			0.20276		0.40552	0.57931
Pirimicarb	kg	0.0825	0.0825	0.16112		0.21724	0.40034
Pronamide	kg			0.059659		0.11932	0.17045
Propaquizafop	kg	0.0165	0.0165	0.027204		0.033409	0.065727
Propiconazole	kg	0.092045	0.092045	0.068182	0.078409		0.017045
Pyraclostrobin (prop)	kg	0.34278	0.34278	0.25391	0.29199		0.063477
Quizalofop ethyl ester	kg	0.007071	0.0070714	0.011659		0.014318	0.028169
Simazine	kg	0.12	0.12	0.12	0.12	0.165	0.12
Spinosad	kg			0.013903		0.027807	0.039724
Sulcotrione	kg	0.072	0.072	0.072	0.072	0.099	0.072
Tepraloxydim	kg	0.007071	0.0070714	0.011659		0.014318	0.028169
Thiacloprid	kg			0.027807		0.055614	0.079448
Thifensulfuron-methyl	kg	0.021996	0.021965	0.017329	0.020027	0.0049	0.007345
Triazamate	kg	0.00616	0.00616	0.00392			0.00672
Triclopyr	kg	0.032178	0.031845	0.032395	0.039746	0.039063	0.033036
Trifloxystrobin	kg	0.13807	0.13807	0.10227	0.11761		0.025568
Trifluralin	kg	0.35345	0.35345	0.53673	0.30109	0.54982	0.85091

Table A3.9: *inventories of all crop farm scenarios for the Broye region, according to ecoinvent and SALCA nomenclature (continued pages 172 to 174).*

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Resources									
Water, river	l	11247108	18351626	24521329	27191329	20250030	19647000	3203600	3661100
Materials/fuels									
[sulfonyl]urea-compounds, at regional storehouse/CH S	kg	5.5904	5.5904	4.6224	5.1965	5.1122	5.2281	4.6224	4.6924
[thio]carbamate-compounds, at regional storehouse/CH S	kg	3.72	3.72	4.2124	3.8211	4.0878	3.7106	4.2124	4.1946
2,4-D, at regional storehouse/CH S	kg	1.1178	1.1178	0.89045	1.0836	1.0195	1.0834	0.89045	0.90888
Acetamide-anillide-compounds, at regional storehouse/CH S	kg	3.2316	3.2316	4.7201	3.3746	4.7201	3.458	4.7201	4.7201
Agricultural machinery, general, production/CH/I S	kg	542.75	542.75	470.98	514.35	395.82	528.02	470.98	460.24
Agricultural machinery, tillage, production/CH/I S	kg	278.24	278.24	285.76	278.53	278	278.01	285.76	284.65
Ammonium nitrate, as N, at regional storehouse/RER S	kg	2336.7	2110.3	1787	1843.4	928.15	2189.7	1548.6	1536.1
Ammonium sulphate, as N, at regional storehouse/RER S	kg	179.75	162.33	137.46	141.8	71.396	168.44	119.12	118.16
Atrazine, at regional storehouse/CH S	kg				0.15636		0.15636		
Baling/CH S	p	0.1289	0.1289	0.1289	0.1289	0.1289	0.1289	0.1289	0.1289
Barley seed IP, at regional storehouse/CH S	kg	1.32	1.32	1.32		3.63		1.32	1.65
Benzimidazole-compounds, at regional storehouse/CH S	kg	6.1779	6.1779	5.0636	5.9272	5.6195	5.8826	5.0636	5.143
Benzo[thia]diazole-compounds, at regional storehouse/CH S	kg	3.3414	3.3414	2.6745	3.2624	2.824	3.2926	2.6745	2.6958
Calcium ammonium nitrate, as N, at regional storehouse/RER S	kg	1168.3	1055.2	893.5	921.71	464.07	1094.8	774.28	768.04
Combine harvesting/CH S	ha	21	21	20.4	20.4	22.5	20.7	20.4	20.7
Copper oxide, at plant/RER S	kg	26.919	26.919	23.554	23.554		26.919	23.554	20.189
Cyclic N-compounds, at regional storehouse/CH S	kg	5.1394	5.1394	4.4911	4.9522	5.0275	4.8707	4.4911	4.5677
Diammonium phosphate, as P2O5, at regional storehouse/RER S	kg	46.468	43.075	36.364	39.563	11.634	43.594	31.95	31.95
Dicamba, at regional storehouse/CH S	kg	0.14585	0.14585	0.11592	0.13604	0.13309	0.13604	0.11592	0.11837
Diesel combustion, in tractor/kg/CH U	kg	3568.1	3568.1	3493.8	3520.2	3266.8	3552.6	3493.8	3461.4
Diesel, at regional storage/CH S	kg	3568.1	3568.1	3493.8	3520.2	3266.8	3552.6	3493.8	3461.4
Dinitroaniline-compounds, at regional storehouse/CH S	kg	5.2112	5.2112	4.7948	5.0793	4.0968	5.3025	4.7948	4.6951
Diphenylether-compounds, at regional storehouse/CH S	kg	0.6055	0.6055	0.70338	0.58491	0.66348	0.59006	0.70338	0.69768
Garage, wood, non-insulated, fire-protected, at farm/m2/CH/I U	m2	0.95628	0.95628	0.94437	0.95562	0.84907	0.9635	0.94437	0.93075
Glyphosate, at regional storehouse/CH S	kg	0.0149	0.0149	0.01592	0.01796	0.01235	0.01745	0.01592	0.01541
Grain drying, low temperature/CH S	kg	2342.8	2342.8	2690.3	6726.2	2868.9	6793.3	2690.3	2715.8
Grass seed IP, at regional storehouse/CH S	kg	1.9125	1.9125	2.04	2.295	1.5937	2.2313	2.04	1.9762
Harvester, production/CH/I S	kg	3.0056	3.0056	3.2096	3.6176	2.4956	3.5156	3.2096	3.1076
Harvesting, by complete harvester, beets/CH S	ha	6.6	6.6	7.5	7.5	7.5	6.9	7.5	7.5
Irrigation pipeline, at farm/m/CH/I U	m	48.75	48.75	52	58.5	40.625	56.875	11.375	9.75
Light fuel oil, burned in boiler 10kW, non-modulating/CH S	MJ	29236	29236	29236	29236	29236	29236	29236	29236
Linuron, at regional storehouse/CH S	kg	0.48951	0.48951	0.40317	0.44642	0.28617	0.46841	0.40317	0.38645
Magnesium oxide, at plant/RER S	kg	122.42	113.81	104.51	106.15	54.75	113.51	80.922	80.922
Maize seed IP, at regional storehouse/CH S	kg				0.3		0.3		
MCPA, at regional storehouse/CH S	kg	2.2834	2.2834	1.8173	2.1322	2.083	2.1319	1.8173	1.8553
Monoammonium phosphate, as P2O5, at regional storehouse/RER S	kg	46.468	43.075	36.364	39.563	11.634	43.594	31.95	31.95

Table A3.9 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Nitrile-compounds, at regional storehouse/CH S	kg	2.6724	2.6724	2.1853	2.4657	2.5778	2.4695	2.1853	2.2414
Organophosphorus-compounds, at regional storehouse/CH S	kg	1.7466	1.7466	2.4503	1.6992	2.4503	1.708	2.4503	2.4503
Passenger car/RER/I S	p	0.040091	0.040091	0.040499	0.040499	0.039071	0.040295	0.040499	0.040295
Pesticide unspecified, at regional storehouse/CH S	kg	1.0164	1.0164	0.99069	0.99108	1.052	0.99553	0.99069	0.99945
Petrol combustion, unleaded, in motor mower/kg/CH U	kg	409.08	409.08	413.89	415.42	397.04	413.01	413.89	411.48
Petrol, unleaded, at regional storage/CH S	kg	409.08	409.08	413.89	415.42	397.04	413.01	413.89	411.48
Phenoxy-compounds, at regional storehouse/CH S	kg	2.7027	2.7027	2.1751	2.5567	2.2643	2.5828	2.1751	2.1878
Phosphate rock, as P2O5, beneficiated, dry, at plant/MA S	kg	79.66	73.844	62.338	67.823	19.944	74.733	54.771	54.771
Potassium chloride, as K2O, at regional storehouse/RER S	kg	494.52	484.92	476.27	500.98	76.14	512.29	441.86	441.86
Potassium sulphate, as K2O, at regional storehouse/RER S	kg	31.565	30.952	30.4	31.978	4.86	32.7	28.204	28.204
Potato haulm cutting/CH S	ha	2.4	2.4	2.1	2.1		2.4	2.1	1.8
Potato seed IP, at regional storehouse/CH S	kg	60.48	60.48	52.92	52.92		60.48	52.92	45.36
Pumping station, concrete walls, non-insulated, at farm/m3/CH/I U	m3	0.00375	0.00375	0.004	0.0045	0.003125	0.004375	0.000875	0.00075
Pyretroid-compounds, at regional storehouse/CH S	kg	0.10121	0.10121	0.14059	0.099316	0.14059	0.099325	0.14059	0.14059
Pyridazine-compounds, at regional storehouse/CH S	kg	1.3483	1.3483	1.5321	1.5321	1.5321	1.4096	1.5321	1.5321
Rape seed IP, at regional storehouse/CH S	kg	0.264	0.264	0.4125	0.231	0.4125	0.2475	0.4125	0.4125
Shed, large, wood, non-insulated, fire-unprotected, at farm/m2/CH/I U	m2	3.9991	3.9991	3.6932	3.8912	3.2544	3.9634	3.6932	3.6305
Single superphosphate, as P2O5, at regional storehouse/RER S	kg	6.6383	6.1536	5.1948	5.6519	1.662	6.2277	4.5642	4.5642
Sowing/CH S	ha	6.6	6.6	7.5	8.7	7.5	8.1	7.5	7.5
Sugar beet seed IP, at regional storehouse/CH S	kg	0.1386	0.1386	0.1575	0.1575	0.1575	0.1449	0.1575	0.1575
Thomas meal, as P2O5, at regional storehouse/RER S	kg	16.596	15.384	12.987	14.13	4.155	15.569	11.411	11.411
Tractor, production/CH/I S	kg	387.4	387.4	378.55	381.86	335.22	387.38	378.55	372.36
Trailer, production/CH/I S	kg	159.68	159.68	159.68	152.68	107.18	162.18	159.68	152.18
Transport, barge/RER S	tkm	12637	11426	9686.7	10007	4954.9	11851	8402.2	8336.7
Transport, freight, rail/CH S	tkm	1538.2	1398.1	1196.5	1239.1	579.4	1450.1	1041.6	1034.4
Transport, lorry 16-32t, EURO3/RER S	tkm	1538.2	1398.1	1196.5	1239.1	579.4	1450.1	1041.6	1034.4
Transport, tractor and trailer/CH S	tkm	4752	4752	5400	5400	5400	4968	5400	5400
Transport, van <3.5t/CH S	tkm	3.6033	3.6033	3.2376	3.3547	1.8106	3.5658	3.2376	3.0338
Triazine-compounds, at regional storehouse/CH S	kg	1.9805	1.9805	2.2125	2.2797	2.1366	2.123	2.2125	2.2017
Triple superphosphate, as P2O5, at regional storehouse/RER S	kg	136.09	126.15	106.49	115.86	34.071	127.67	93.567	93.567
Urea, as N, at regional storehouse/RER S	kg	808.85	730.49	618.58	638.11	321.28	757.96	536.04	531.72
Wheat seed IP, at regional storehouse/CH S	kg	27	27	21.06	27	21.06	27	21.06	21.06
Emissions to air (low population)									
Ammonia	kg	249.91	225.7	191.12	197.15	99.266	234.19	165.62	164.28
Carbon dioxide, fossil	kg	1269.9	1146.9	971.17	1001.8	504.41	1190	841.58	834.8
Dinitrogen monoxide	kg	106.24	99.312	88.917	93.859	61.176	103.43	81.327	80.848
Methane, biogenic	kg	4.0893	4.0893	4.0893	4.0893	4.0893	4.0893	4.0893	4.0893
Nitrogen oxides	kg	22.31	20.855	18.673	19.71	12.847	21.721	17.079	16.978
Emissions to water									
Cadmium, ion, groundwater	kg	0.000785	0.0007561	0.000693	0.000724	0.000328	0.000761	0.000645	0.000645
Cadmium, ion, river	kg	1.56E-13	1.505E-13	1.26E-13	1.63E-13	7.95E-14	1.67E-13	1.17E-13	1.17E-13
Chromium, ion, groundwater	kg	0.50494	0.49675	0.47849	0.48714	0.32659	0.49833	0.46178	0.46168
Chromium, ion, river	kg	1.84E-11	1.811E-11	1.59E-11	2E-11	1.45E-11	2.01E-11	1.54E-11	1.54E-11
Copper, ion, groundwater	kg	0.069122	0.066921	0.062988	0.064218	0.040587	0.067732	0.059487	0.059348
Copper, ion, river	kg	1.13E-11	1.096E-11	9.42E-12	1.19E-11	8.09E-12	1.23E-11	8.89E-12	8.87E-12

Table A3.9 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Lead, groundwater	kg	0.00119	0.0010986	0.000958	0.001	0.000418	0.001134	0.000846	0.000842
Lead, river	kg	1.57E-12	1.451E-12	1.15E-12	1.49E-12	6.73E-13	1.65E-12	1.02E-12	1.02E-12
Mercury, groundwater	kg	5.13E-08	5.126E-08	4.73E-08	4.73E-08	2.4E-08	5.06E-08	4.73E-08	4.4E-08
Mercury, river	kg	1.12E-16	1.118E-16	9.41E-17	1.16E-16	6.38E-17	1.22E-16	9.41E-17	8.75E-17
Nickel, ion, river	kg	1.06E-11	1.012E-11	8.44E-12	1.07E-11	6.95E-12	1.14E-11	7.78E-12	7.75E-12
Nitrate, groundwater	kg	3402.8	3488.9	2979.2	3411.3	2805.1	3450.3	2958.8	2934.4
Phosphate, groundwater	kg	24.366	24.671	25.834	29.019	20.498	28.318	25.8	24.92
Phosphorus, river	kg	0.001245	0.0012561	0.001338	0.001735	0.00112	0.00167	0.001336	0.0013
Zinc, ion, groundwater	kg	0.24677	0.23021	0.20336	0.21074	0.1035	0.23588	0.18231	0.18147
Zinc, ion, river	kg	1.56E-11	1.452E-11	1.17E-11	1.5E-11	7.94E-12	1.64E-11	1.05E-11	1.04E-11
Emissions to soil									
2,4-D	kg	1.1178	1.1178	0.89045	1.0836	1.0195	1.0834	0.89045	0.90888
Acetamiprid	kg	0.009103	0.0091034	0.014224	0.007966	0.014224	0.008535	0.014224	0.014224
Acibenzolar-s-methyl	kg	0.040909	0.040909	0.031909	0.040909	0.031909	0.040909	0.031909	0.031909
Aclonifen	kg	0.19636	0.19636	0.17182	0.17182		0.19636	0.17182	0.14727
Alachlor	kg				0.1008		0.1008		
Alpha-cypermethrin	kg	0.010855	0.010855	0.015115	0.010631	0.015115	0.010645	0.015115	0.015115
Amidosulfuron	kg	0.023885	0.023885	0.019103	0.022397	0.02176	0.022382	0.019103	0.019482
Asulam	kg	0.039733	0.039733	0.042453	0.047893	0.032933	0.046533	0.042453	0.041093
Atrazine	kg				0.15636		0.15636		
Azoxystrobin	kg	0.52668	0.52668	0.42168	0.47727	0.50815	0.47727	0.42168	0.43404
Bentazone	kg	3.3005	3.3005	2.6426	3.2215	2.7921	3.2516	2.6426	2.6639
Bifenthrin	kg	0.008817	0.0088165	0.012276	0.008635	0.012276	0.008646	0.012276	0.012276
Bromoxynil	kg	0.47369	0.47369	0.37649	0.46702	0.43226	0.46702	0.37649	0.38445
Carbetamide	kg	0.43636	0.43636	0.68182	0.38182	0.68182	0.40909	0.68182	0.68182
Carbofuran	kg	0.22	0.22	0.25	0.25	0.25	0.23	0.25	0.25
Carfentrazone-ethyl	kg	0.03655	0.03655	0.02905	0.034091	0.033353	0.034091	0.02905	0.029665
Chloridazon	kg	1.3483	1.3483	1.5321	1.5321	1.5321	1.4096	1.5321	1.5321
Chlorothalonil	kg	5.6869	5.6869	4.5057	5.3693	5.0616	5.3693	4.5057	4.5851
Chlorotoluron	kg	1.3743	1.3743	1.0923	1.2818	1.2541	1.2818	1.0923	1.1154
Chlorpyrifos	kg	0.66124	0.66124	0.92069	0.64758	0.92069	0.64841	0.92069	0.92069
Clethodim	kg	0.14643	0.14643	0.18187	0.14587	0.15896	0.14624	0.18187	0.1786
Clomazone	kg	0.083389	0.083389	0.13029	0.072965	0.13029	0.078177	0.13029	0.13029
Clopyralid	kg	0.041982	0.041982	0.047673	0.047741	0.047554	0.043953	0.047673	0.047656
Copper	kg	26.919	26.919	23.554	23.554		26.919	23.554	20.189
Cycloxydim	kg	0.14384	0.14384	0.18565	0.14965	0.16656	0.15132	0.18565	0.18292
Cypermethrin	kg	0.060614	0.060614	0.084396	0.059362	0.084396	0.059438	0.084396	0.084396
Cyproconazole	kg	0.24069	0.24069	0.19271	0.21811	0.23223	0.21811	0.19271	0.19835
Cyprodinil	kg	0.90288	0.90288	0.72288	0.81818	0.87112	0.81818	0.72288	0.74406
Deltamethrin	kg	0.017751	0.017751	0.024319	0.017628	0.024319	0.017508	0.024319	0.024319
Desmedipham	kg	0.092683	0.092683	0.10532	0.10532	0.10532	0.096895	0.10532	0.10532
Diazinon	kg	0.38572	0.38572	0.53707	0.37776	0.53707	0.37824	0.53707	0.53707
Dicamba	kg	0.14585	0.14585	0.11592	0.13604	0.13309	0.13604	0.11592	0.11837
Dichlobenil	kg	0.005588	0.0055875	0.00597	0.006735	0.004631	0.006544	0.00597	0.005779
Difenoconazole	kg	0.51136	0.51136	0.39886	0.51136	0.39886	0.51136	0.39886	0.39886
Diflufenican	kg	0.15863	0.15863	0.12608	0.14795	0.14475	0.14795	0.12608	0.12874
Dimefuron	kg	0.21818	0.21818	0.34091	0.19091	0.34091	0.20454	0.34091	0.34091
Dimethenamid	kg	0.3564	0.3564	0.405	0.5541	0.405	0.5217	0.405	0.405
Dimethoate	kg	0.088	0.088	0.1	0.1	0.1	0.092	0.1	0.1
Epoxiconazole	kg	1.3282	1.3282	1.0647	1.1974	1.2935	1.1974	1.0647	1.0974
Ethofumesate	kg	0.49094	0.49094	0.55789	0.55789	0.55789	0.51326	0.55789	0.55789
Famoxadone	kg	0.45144	0.45144	0.36144	0.40909	0.43556	0.40909	0.36144	0.37203
Fenpropimorph	kg	1.0604	1.0604	0.85042	0.95454	1.0357	0.95454	0.85042	0.87689
Fluazifop	kg	0.26971	0.26971	0.34809	0.26372	0.3123	0.26686	0.34809	0.34298
Fluroxypyr	kg	0.27724	0.27724	0.22035	0.25858	0.25299	0.25858	0.22035	0.22501
Flusilazole	kg	0.48214	0.48214	0.38602	0.43691	0.46518	0.43691	0.38602	0.39733
Glyphosate	kg	0.0149	0.0149	0.01592	0.01796	0.01235	0.01745	0.01592	0.01541
Haloxyfop-P-methyl	kg	0.051784	0.051784	0.066834	0.050634	0.059961	0.051237	0.066834	0.065852
Heptenophos	kg	0.07117	0.07117	0.080875	0.080875	0.080875	0.074405	0.080875	0.080875
Hexaconazole	kg	0.25568	0.25568	0.19943	0.25568	0.19943	0.25568	0.19943	0.19943

Table A3.9 continued

Flow	Unit	Scenario							
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050	Ext2050, water pricing	Ext2050, water quota
Ioxynil	kg	0.24854	0.24854	0.19754	0.23182	0.2268	0.23182	0.19754	0.20172
Isoproturon	kg	3.1287	3.1287	2.4867	2.9182	2.855	2.9182	2.4867	2.5393
Kresoxim-methyl	kg	0.3762	0.3762	0.3012	0.34091	0.36297	0.34091	0.3012	0.31003
Lambda-cyhalothrin	kg	0.008679	0.0086793	0.012155	0.008457	0.012155	0.008493	0.012155	0.012155
Lead	kg	0.020566	0.018783	0.01339	0.016612	0.005514	0.019514	0.011647	0.01159
Lenacil	kg	0.90514	0.90514	1.0286	1.0286	1.0286	0.94628	1.0286	1.0286
Linuron	kg	0.48951	0.48951	0.40317	0.44642	0.28617	0.46841	0.40317	0.38645
MCPA	kg	2.2834	2.2834	1.8173	2.1322	2.083	2.1319	1.8173	1.8553
MCPB	kg	0.23308	0.23308	0.20683	0.20887	0.01235	0.23563	0.20683	0.17905
Mecoprop-P	kg	1.0745	1.0745	0.85745	1.0056	0.97949	1.0052	0.85745	0.87488
Metamitron	kg	1.8482	1.8482	2.1002	2.1002	2.1002	1.9322	2.1002	2.1002
Metazachlor	kg	1.0403	1.0403	1.6254	0.91023	1.6254	0.97525	1.6254	1.6254
Methidathion	kg	0.077	0.077	0.0875	0.0875	0.0875	0.0805	0.0875	0.0875
Methomyl	kg	0.11	0.11	0.125	0.125	0.125	0.115	0.125	0.125
Metolachlor, (S)	kg	0.41636	0.41636	0.47314	0.56818	0.47314	0.53033	0.47314	0.47314
Metoxuron	kg	0.70176	0.70176	0.55776	0.65454	0.64038	0.65454	0.55776	0.56956
Metribuzin	kg	0.090622	0.090622	0.079294	0.079294		0.090622	0.079294	0.067966
Metsulfuron-methyl	kg	0.006423	0.0064227	0.005088	0.006065	0.005714	0.006065	0.005088	0.005178
Monolinuron	kg	0.14364	0.14364	0.12568	0.12568		0.14364	0.12568	0.10773
Napropamide	kg	1.2822	1.2822	2.0035	1.122	2.0035	1.2021	2.0035	2.0035
Nickel	kg	0.050376	0.041027	0.016103	0.023181		0.0395	0.008438	0.007956
Nicosulfuron	kg				0.003		0.003		
Orbencarb	kg	0.26293	0.26293	0.23006	0.23006		0.26293	0.23006	0.1972
Pendimethalin	kg	3.6847	3.6847	3.0834	3.7031	2.4746	3.8625	3.0834	2.9964
Phenmedipham	kg	1.4445	1.4445	1.6415	1.6415	1.6415	1.5101	1.6415	1.6415
Phosalone	kg	0.46345	0.46345	0.72414	0.40552	0.72414	0.43448	0.72414	0.72414
Pirimicarb	kg	0.41327	0.41327	0.57543	0.40474	0.57543	0.40526	0.57543	0.57543
Pronamide	kg	0.13636	0.13636	0.21307	0.11932	0.21307	0.12784	0.21307	0.21307
Propaquizafop	kg	0.083909	0.083909	0.1083	0.082045	0.097159	0.083022	0.1083	0.1067
Propiconazole	kg	0.17045	0.17045	0.13295	0.17045	0.13295	0.17045	0.13295	0.13295
Pyraclostrobin (prop)	kg	0.70049	0.70049	0.56084	0.63477	0.67584	0.63477	0.56084	0.57727
Quizalofop ethyl ester	kg	0.035961	0.035961	0.046412	0.035162	0.041639	0.035581	0.046412	0.04573
Simazine	kg				0.06		0.06		
Spinosad	kg	0.031779	0.031779	0.049655	0.027807	0.049655	0.029793	0.049655	0.049655
Sulcotrione	kg				0.036		0.036		
Tepraloxydim	kg	0.035961	0.035961	0.046412	0.035162	0.041639	0.035581	0.046412	0.04573
Thiacloprid	kg	0.063558	0.063558	0.09931	0.055614	0.09931	0.059586	0.09931	0.09931
Thifensulfuron-methyl	kg	0.035276	0.035276	0.027923	0.034154	0.030696	0.034148	0.027923	0.028319
Triazamate	kg	0.01232	0.01232	0.014	0.014	0.014	0.01288	0.014	0.014
Triclopyr	kg	0.001987	0.0019867	0.002123	0.002395	0.001647	0.002327	0.002123	0.002055
Trifloxystrobin	kg	0.28215	0.28215	0.2259	0.25568	0.27222	0.25568	0.2259	0.23252
Trifluralin	kg	1.3301	1.3301	1.5396	1.2044	1.6222	1.2436	1.5396	1.5514

Table A3.10: *inventories of all crop farm scenarios for the Greifensee region, according to ecoinvent and SALCA nomenclature (continued pages 176 to 179).*

		Scenario					
Flow	Unit	Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Resources							
Water, river	l	1713902	2022132	10497026	10497026	7875000	19647000
Materials/fuels							
[sulfonyl]urea-compounds, at regional storehouse/CH S	kg	5.5904	5.5904	4.6224	5.1965	5.1122	5.2281
[thio]carbamate-compounds, at regional storehouse/CH S	kg	3.72	3.72	4.2124	3.8211	4.0878	3.7106
2,4-D, at regional storehouse/CH S	kg	1.1178	1.1178	0.89045	1.0836	1.0195	1.0834
Acetamide-anillide-compounds, at regional storehouse/CH S	kg	3.2316	3.2316	4.7201	3.3746	4.7201	3.458
Agricultural machinery, general, production/CH/I S	kg	542.75	542.75	470.98	514.35	395.82	528.02
Agricultural machinery, tillage, production/CH/I S	kg	278.24	278.24	285.76	278.53	278	278.01
Ammonium nitrate, as N, at regional storehouse/RER S	kg	2336.7	2110.3	1787	1843.4	928.15	2189.7
Ammonium sulphate, as N, at regional storehouse/RER S	kg	179.75	162.33	137.46	141.8	71.396	168.44
Atrazine, at regional storehouse/CH S	kg						0.15636
Baling/CH S	p	0.1289	0.1289	0.1289	0.1289	0.1289	0.1289
Barley seed IP, at regional storehouse/CH S	kg	1.32	1.32	1.32		3.63	
Benzimidazole-compounds, at regional storehouse/CH S	kg	6.1779	6.1779	5.0636	5.9272	5.6195	5.8826
Benzo[thia]diazole-compounds, at regional storehouse/CH S	kg	3.3414	3.3414	2.6745	3.2624	2.824	3.2926
Calcium ammonium nitrate, as N, at regional storehouse/RER S	kg	1168.3	1055.2	893.5	921.71	464.07	1094.8
Combine harvesting/CH S	ha	21	21	20.4	20.4	22.5	20.7
Copper oxide, at plant/RER S	kg	26.919	26.919	23.554	23.554		26.919
Cyclic N-compounds, at regional storehouse/CH S	kg	5.1394	5.1394	4.4911	4.9522	5.0275	4.8707
Diammonium phosphate, as P2O5, at regional storehouse/RER S	kg	46.468	43.075	36.364	39.563	11.634	43.594
Dicamba, at regional storehouse/CH S	kg	0.14585	0.14585	0.11592	0.13604	0.13309	0.13604
Diesel combustion, in tractor/kg/CH U	kg	3568.1	3568.1	3493.8	3520.2	3266.8	3552.6
Diesel, at regional storage/CH S	kg	3568.1	3568.1	3493.8	3520.2	3266.8	3552.6
Dinitroaniline-compounds, at regional storehouse/CH S	kg	5.2112	5.2112	4.7948	5.0793	4.0968	5.3025
Diphenylether-compounds, at regional storehouse/CH S	kg	0.6055	0.6055	0.70338	0.58491	0.66348	0.59006
Garage, wood, non-insulated, fire-protected, at farm/m2/CH/I U	m2	0.95628	0.95628	0.94437	0.95562	0.84907	0.9635
Glyphosate, at regional storehouse/CH S	kg	0.0149	0.0149	0.01592	0.01796	0.01235	0.01745
Grain drying, low temperature/CH S	kg	2342.8	2342.8	2690.3	6726.2	2868.9	6793.3
Grass seed IP, at regional storehouse/CH S	kg	1.9125	1.9125	2.04	2.295	1.5937	2.2313
Harvester, production/CH/I S	kg	3.0056	3.0056	3.2096	3.6176	2.4956	3.5156
Harvesting, by complete harvester, beets/CH S	ha	6.6	6.6	7.5	7.5	7.5	6.9
Irrigation pipeline, at farm/m/CH/I U	m	48.75	48.75	52	58.5	40.625	56.875

Table A3.10 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Light fuel oil, burned in boiler 10kW, non-modulating/CH S	MJ	29236	29236	29236	29236	29236	29236
Linuron, at regional storehouse/CH S	kg	0.48951	0.48951	0.40317	0.44642	0.28617	0.46841
Magnesium oxide, at plant/RER S	kg	122.42	113.81	104.51	106.15	54.75	113.51
Maize seed IP, at regional storehouse/CH S					0.3		0.3
MCPA, at regional storehouse/CH S	kg	2.2834	2.2834	1.8173	2.1322	2.083	2.1319
Monoammonium phosphate, as P2O5, at regional storehouse/RER S	kg	46.468	43.075	36.364	39.563	11.634	43.594
Nitrile-compounds, at regional storehouse/CH S	kg	2.6724	2.6724	2.1853	2.4657	2.5778	2.4695
Organophosphorus-compounds, at regional storehouse/CH S	kg	1.7466	1.7466	2.4503	1.6992	2.4503	1.708
Passenger car/RER/I S	p	0.040091	0.040091	0.040499	0.040499	0.039071	0.040295
Pesticide unspecified, at regional storehouse/CH S	kg	1.0164	1.0164	0.99069	0.99108	1.052	0.99553
Petrol combustion, unleaded, in motor mower/kg/CH U	kg	409.08	409.08	413.89	415.42	397.04	413.01
Petrol, unleaded, at regional storage/CH S	kg	409.08	409.08	413.89	415.42	397.04	413.01
Phenoxy-compounds, at regional storehouse/CH S	kg	2.7027	2.7027	2.1751	2.5567	2.2643	2.5828
Phosphate rock, as P2O5, beneficiated, dry, at plant/MA S	kg	79.66	73.844	62.338	67.823	19.944	74.733
Potassium chloride, as K2O, at regional storehouse/RER S	kg	494.52	484.92	476.27	500.98	76.14	512.29
Potassium sulphate, as K2O, at regional storehouse/RER S	kg	31.565	30.952	30.4	31.978	4.86	32.7
Potato haulm cutting/CH S	ha	2.4	2.4	2.1	2.1		2.4
Potato seed IP, at regional storehouse/CH S	kg	60.48	60.48	52.92	52.92		60.48
Pumping station, concrete walls, non-insulated, at farm/m3/CH/I U	m3	0.00375	0.00375	0.004	0.0045	0.003125	0.004375
Pyretroid-compounds, at regional storehouse/CH S	kg	0.10121	0.10121	0.14059	0.099316	0.14059	0.099325
Pyridazine-compounds, at regional storehouse/CH S	kg	1.3483	1.3483	1.5321	1.5321	1.5321	1.4096
Rape seed IP, at regional storehouse/CH S	kg	0.264	0.264	0.4125	0.231	0.4125	0.2475
Shed, large, wood, non-insulated, fire-unprotected, at farm/m2/CH/I U	m2	3.9991	3.9991	3.6932	3.8912	3.2544	3.9634
Single superphosphate, as P2O5, at regional storehouse/RER S	kg	6.6383	6.1536	5.1948	5.6519	1.662	6.2277
Sowing/CH S	ha	6.6	6.6	7.5	8.7	7.5	8.1
Sugar beet seed IP, at regional storehouse/CH S	kg	0.1386	0.1386	0.1575	0.1575	0.1575	0.1449
Thomas meal, as P2O5, at regional storehouse/RER S	kg	16.596	15.384	12.987	14.13	4.155	15.569
Tractor, production/CH/I S	kg	387.4	387.4	378.55	381.86	335.22	387.38
Trailer, production/CH/I S	kg	159.68	159.68	159.68	152.68	107.18	162.18
Transport, barge/RER S	tkm	12637	11426	9686.7	10007	4954.9	11851
Transport, freight, rail/CH S	tkm	1538.2	1398.1	1196.5	1239.1	579.4	1450.1
Transport, lorry 16-32t, EURO3/RER S	tkm	1538.2	1398.1	1196.5	1239.1	579.4	1450.1
Transport, tractor and trailer/CH S	tkm	4752	4752	5400	5400	5400	4968
Transport, van <3.5t/CH S	tkm	3.6033	3.6033	3.2376	3.3547	1.8106	3.5658
Triazine-compounds, at regional storehouse/CH S	kg	1.9805	1.9805	2.2125	2.2797	2.1366	2.123
Triple superphosphate, as P2O5, at regional storehouse/RER S	kg	136.09	126.15	106.49	115.86	34.071	127.67
Urea, as N, at regional storehouse/RER S	kg	808.85	730.49	618.58	638.11	321.28	757.96
Wheat seed IP, at regional storehouse/CH S	kg	27	27	21.06	27	21.06	27

Table A3.10 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Emissions to air (low population)							
Ammonia	kg	249.91	225.7	191.12	197.15	99.266	234.19
Carbon dioxide, fossil	kg	1269.9	1146.9	971.17	1001.8	504.41	1190
Dinitrogen monoxide	kg	106.24	99.312	88.917	93.859	61.176	103.43
Methane, biogenic	kg	4.0893	4.0893	4.0893	4.0893	4.0893	4.0893
Nitrogen oxides	kg	22.31	20.855	18.673	19.71	12.847	21.721
Emissions to water							
Cadmium, ion, groundwater	kg	0.000785	0.0007561	0.000693	0.000724	0.000328	0.000761
Cadmium, ion, river	kg	1.56E-13	1.505E-13	1.26E-13	1.63E-13	7.95E-14	1.67E-13
Chromium, ion, groundwater	kg	0.50494	0.49675	0.47849	0.48714	0.32659	0.49833
Chromium, ion, river	kg	1.84E-11	1.811E-11	1.59E-11	2E-11	1.45E-11	2.01E-11
Copper, ion, groundwater	kg	0.069122	0.066921	0.062988	0.064218	0.040587	0.067732
Copper, ion, river	kg	1.13E-11	1.096E-11	9.42E-12	1.19E-11	8.09E-12	1.23E-11
Lead, groundwater	kg	0.00119	0.0010986	0.000958	0.001	0.000418	0.001134
Lead, river	kg	1.57E-12	1.451E-12	1.15E-12	1.49E-12	6.73E-13	1.65E-12
Mercury, groundwater	kg	5.13E-08	5.126E-08	4.73E-08	4.73E-08	2.4E-08	5.06E-08
Mercury, river	kg	1.12E-16	1.118E-16	9.41E-17	1.16E-16	6.38E-17	1.22E-16
Nickel, ion, river	kg	1.06E-11	1.012E-11	8.44E-12	1.07E-11	6.95E-12	1.14E-11
Nitrate, groundwater	kg	3402.8	3488.9	2979.2	3411.3	2805.1	3450.3
Phosphate, groundwater	kg	24.366	24.671	25.834	29.019	20.498	28.318
Phosphorus, river	kg	0.001245	0.0012561	0.001338	0.001735	0.00112	0.00167
Zinc, ion, groundwater	kg	0.24677	0.23021	0.20336	0.21074	0.1035	0.23588
Zinc, ion, river	kg	1.56E-11	1.452E-11	1.17E-11	1.5E-11	7.94E-12	1.64E-11
Emissions to soil (agricultural)							
2,4-D	kg	1.1178	1.1178	0.89045	1.0836	1.0195	1.0834
Acetamiprid	kg	0.009103	0.0091034	0.014224	0.007966	0.014224	0.008535
Acibenzolar-s-methyl	kg	0.040909	0.040909	0.031909	0.040909	0.031909	0.040909
Aclonifen	kg	0.19636	0.19636	0.17182	0.17182		0.19636
Alachlor	kg				0.1008		0.1008
Alpha-cypermethrin	kg	0.010855	0.010855	0.015115	0.010631	0.015115	0.010645
Amidosulfuron	kg	0.023885	0.023885	0.019103	0.022397	0.02176	0.022382
Asulam	kg	0.039733	0.039733	0.042453	0.047893	0.032933	0.046533
Atrazine	kg				0.15636		0.15636
Azoxystrobin	kg	0.52668	0.52668	0.42168	0.47727	0.50815	0.47727
Bentazone	kg	3.3005	3.3005	2.6426	3.2215	2.7921	3.2516
Bifenthrin	kg	0.008817	0.0088165	0.012276	0.008635	0.012276	0.008646
Bromoxynil	kg	0.47369	0.47369	0.37649	0.46702	0.43226	0.46702
Carbetamide	kg	0.43636	0.43636	0.68182	0.38182	0.68182	0.40909
Carbofuran	kg	0.22	0.22	0.25	0.25	0.25	0.23
Carfentrazone-ethyl	kg	0.03655	0.03655	0.02905	0.034091	0.033353	0.034091
Chloridazon	kg	1.3483	1.3483	1.5321	1.5321	1.5321	1.4096
Chlorothalonil	kg	5.6869	5.6869	4.5057	5.3693	5.0616	5.3693
Chlorotoluron	kg	1.3743	1.3743	1.0923	1.2818	1.2541	1.2818
Chlorpyrifos	kg	0.66124	0.66124	0.92069	0.64758	0.92069	0.64841
Clethodim	kg	0.14643	0.14643	0.18187	0.14587	0.15896	0.14624
Clomazone	kg	0.083389	0.083389	0.13029	0.072965	0.13029	0.078177
Clopyralid	kg	0.041982	0.041982	0.047673	0.047741	0.047554	0.043953
Copper	kg	26.919	26.919	23.554	23.554		26.919
Cycloxydim	kg	0.14384	0.14384	0.18565	0.14965	0.16656	0.15132
Cypermethrin	kg	0.060614	0.060614	0.084396	0.059362	0.084396	0.059438
Cyproconazole	kg	0.24069	0.24069	0.19271	0.21811	0.23223	0.21811
Cyprodinil	kg	0.90288	0.90288	0.72288	0.81818	0.87112	0.81818

Table A3.10 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Deltamethrin	kg	0.017751	0.017751	0.024319	0.017628	0.024319	0.017508
Desmedipham	kg	0.092683	0.092683	0.10532	0.10532	0.10532	0.096895
Diazinon	kg	0.38572	0.38572	0.53707	0.37776	0.53707	0.37824
Dicamba	kg	0.14585	0.14585	0.11592	0.13604	0.13309	0.13604
Dichlobenil	kg	0.005588	0.0055875	0.00597	0.006735	0.004631	0.006544
Difenoconazole	kg	0.51136	0.51136	0.39886	0.51136	0.39886	0.51136
Diflufenican	kg	0.15863	0.15863	0.12608	0.14795	0.14475	0.14795
Dimefuron	kg	0.21818	0.21818	0.34091	0.19091	0.34091	0.20454
Dimethenamid	kg	0.3564	0.3564	0.405	0.5541	0.405	0.5217
Dimethoate	kg	0.088	0.088	0.1	0.1	0.1	0.092
Epoxiconazole	kg	1.3282	1.3282	1.0647	1.1974	1.2935	1.1974
Ethofumesate	kg	0.49094	0.49094	0.55789	0.55789	0.55789	0.51326
Famoxadone	kg	0.45144	0.45144	0.36144	0.40909	0.43556	0.40909
Fenpropimorph	kg	1.0604	1.0604	0.85042	0.95454	1.0357	0.95454
Fluazifop	kg	0.26971	0.26971	0.34809	0.26372	0.3123	0.26686
Fluroxypyr	kg	0.27724	0.27724	0.22035	0.25858	0.25299	0.25858
Flusilazole	kg	0.48214	0.48214	0.38602	0.43691	0.46518	0.43691
Glyphosate	kg	0.0149	0.0149	0.01592	0.01796	0.01235	0.01745
Haloxyfop-P-methyl	kg	0.051784	0.051784	0.066834	0.050634	0.059961	0.051237
Heptenophos	kg	0.07117	0.07117	0.080875	0.080875	0.080875	0.074405
Hexaconazole	kg	0.25568	0.25568	0.19943	0.25568	0.19943	0.25568
Ioxynil	kg	0.24854	0.24854	0.19754	0.23182	0.2268	0.23182
Isoproturon	kg	3.1287	3.1287	2.4867	2.9182	2.855	2.9182
Kresoxim-methyl	kg	0.3762	0.3762	0.3012	0.34091	0.36297	0.34091
Lambda-cyhalothrin	kg	0.008679	0.0086793	0.012155	0.008457	0.012155	0.008493
Lead	kg	0.020566	0.018783	0.01339	0.016612	0.005514	0.019514
Lenacil	kg	0.90514	0.90514	1.0286	1.0286	1.0286	0.94628
Linuron	kg	0.48951	0.48951	0.40317	0.44642	0.28617	0.46841
MCPA	kg	2.2834	2.2834	1.8173	2.1322	2.083	2.1319
MCPB	kg	0.23308	0.23308	0.20683	0.20887	0.01235	0.23563
Mecoprop-P	kg	1.0745	1.0745	0.85745	1.0056	0.97949	1.0052
Metamitron	kg	1.8482	1.8482	2.1002	2.1002	2.1002	1.9322
Metazachlor	kg	1.0403	1.0403	1.6254	0.91023	1.6254	0.97525
Methidathion	kg	0.077	0.077	0.0875	0.0875	0.0875	0.0805
Methomyl	kg	0.11	0.11	0.125	0.125	0.125	0.115
Metolachlor, (S)	kg	0.41636	0.41636	0.47314	0.56818	0.47314	0.53033
Metoxuron	kg	0.70176	0.70176	0.55776	0.65454	0.64038	0.65454
Metribuzin	kg	0.090622	0.090622	0.079294	0.079294		0.090622
Metsulfuron-methyl	kg	0.006423	0.0064227	0.005088	0.006065	0.005714	0.006065
Monolinuron	kg	0.14364	0.14364	0.12568	0.12568		0.14364
Napropamide	kg	1.2822	1.2822	2.0035	1.122	2.0035	1.2021
Nickel	kg	0.050376	0.041027	0.016103	0.023181		0.0395
Nicosulfuron	kg				0.003		0.003
Orbencarb	kg	0.26293	0.26293	0.23006	0.23006		0.26293
Pendimethalin	kg	3.6847	3.6847	3.0834	3.7031	2.4746	3.8625
Phenmedipham	kg	1.4445	1.4445	1.6415	1.6415	1.6415	1.5101
Phosalone	kg	0.46345	0.46345	0.72414	0.40552	0.72414	0.43448
Pirimicarb	kg	0.41327	0.41327	0.57543	0.40474	0.57543	0.40526
Pronamide	kg	0.13636	0.13636	0.21307	0.11932	0.21307	0.12784
Propaquizafop	kg	0.083909	0.083909	0.1083	0.082045	0.097159	0.083022
Propiconazole	kg	0.17045	0.17045	0.13295	0.17045	0.13295	0.17045
Pyraclostrobin (prop)	kg	0.70049	0.70049	0.56084	0.63477	0.67584	0.63477

Table A3.10 continued

Flow	Unit	Scenario					
		Reference	Ext2050 without adaptation	Ext2050	Ext2050, subsidy change	Ext2050, european prices	Mod2050
Quizalofop ethyl ester	kg	0.035961	0.035961	0.046412	0.035162	0.041639	0.035581
Simazine	kg				0.06		0.06
Spinosad	kg	0.031779	0.031779	0.049655	0.027807	0.049655	0.029793
Sulcotrione	kg				0.036		0.036
Tepraloxydim	kg	0.035961	0.035961	0.046412	0.035162	0.041639	0.035581
Thiacloprid	kg	0.063558	0.063558	0.09931	0.055614	0.09931	0.059586
Thifensulfuron-methyl	kg	0.035276	0.035276	0.027923	0.034154	0.030696	0.034148
Triazamate	kg	0.01232	0.01232	0.014	0.014	0.014	0.01288
Triclopyr	kg	0.001987	0.0019867	0.002123	0.002395	0.001647	0.002327
Trifloxystrobin	kg	0.28215	0.28215	0.2259	0.25568	0.27222	0.25568
Trifluralin	kg	1.3301	1.3301	1.5396	1.2044	1.6222	1.2436

ANNEX 4: LIFE CYCLE ASSESSMENT OF REGIONAL ADAPTATION SCENARIOS

A4.1. Description of regional optimization model

The regional agricultural scenarios were developed using a combination of the crop growth model CropSyst [217] to simulate crop yields based on management variables and environmental conditions, and a simple livestock model. The latter simulates the water and pasture area requirements of the livestock (no other constraints between the two modules are considered). The model is spatially explicit: the region is modeled as a grid of 1709 pixels of 25 ha. Land use and management is defined for each pixel according to its environmental characteristics (air temperature, precipitation, slope, soil type), in order to optimize the goals for the whole region. The optimization goals are multiple: maximization of productivity (measured as an index of production, grouping all products (in kg) relative to the minimum and maximum achieved in the region), and minimization of soil erosion, nitrate leaching and irrigation water use. These goals can be weighted relative to each other, reflecting possible priorities the region may set. The spatially-explicit optimization variables are crop rotation choice and surface allocation, fertilization intensity level, irrigation intensity, tillage strategy (conventional or conservation tillage) and stocking density. The available crop rotations consist of 50 realistic rotations for the region, composed of the main crops in the region (winter wheat, winter barley, grain maize, winter rapeseed, potatoes, sugar beet, silo maize, grass), as well as permanent grasslands and pastures. Note that the number and type of animals are not a variable here, and remain fixed at the current levels for all scenarios. Further elements which are not considered as variables include pest control strategy and irrigation technology, for similar reasons as in the farm optimization model: the focus was set on large-scale strategic adaptation measures rather than incremental technical improvements; and the prediction of pest control strategy in 2050 was considered too uncertain to include in the models. Economic considerations are also not included in this modeling approach.

A4.2. Absolute impacts and functional units of all scenarios

The absolute impacts (not related to a functional unit) and the functional units of all impact categories and all scenarios at the regional scale are listed in Table A4.1.

Table A4.1: *absolute impacts (initial indicator selection) and functional units for all scenarios assessed at the regional scale.*

	Unit	Reference	Ext2050 without adaptation	Ext2050, productivity	Ext2050, preservation	Ext2050, compromise
Impact category						
Non-renewable energy	MJ eq.	5.96E+09	5.96E+09	6.29E+09	4.57E+09	5.70E+09
GWP	kg CO_2 eq.	1.08E+09	1.09E+09	9.37E+08	8.16E+08	9.89E+08
Ozone formation	m^2*ppm*h	6.67E+09	6.69E+09	6.13E+09	5.29E+09	6.08E+09
Acidification	m^2	2.93E+08	2.93E+08	2.43E+08	2.76E+08	2.62E+08
Terrestrial eutrophication	m^2	1.17E+09	1.17E+09	9.52E+08	1.12E+09	1.04E+09
Freshwater eutrophication	kg P eq.	1.16E+06	1.42E+06	1.91E+06	4.64E+05	7.45E+05
Marine eutrophication	kg N eq.	1.05E+07	1.34E+07	1.54E+07	7.83E+06	1.03E+07
Human toxicity	kg 1,4-DB eq.	3.33E+08	3.33E+08	3.46E+08	2.62E+08	3.04E+08
Terrestrial ecotoxicity	kg 1,4-DB eq.	3.80E+06	3.45E+06	1.55E+06	4.06E+06	3.67E+06
Aquatic ecotoxicity	kg 1,4-DB eq.	2.04E+08	2.24E+08	2.40E+08	1.18E+08	1.50E+08
Land competition	m^2*y	2.85E+09	2.85E+09	2.99E+09	3.21E+09	2.88E+09
Aquatic biodiversity loss	GSE*y	4.16E-07	6.22E-07	1.81E-05	0	4.73E-07
Reduction in terrestrial biodiversity	points lost	8.48E+00	8.48E+00	1.41E+01	7.02E+00	1.08E+01
Functional unit						
Production	MJ dig. en.	9.38E+09	9.01E+09	1.16E+10	4.54E+09	6.21E+09
Land occupation	ha*y	2.14E+05	2.14E+05	2.14E+05	2.14E+05	2.14E+05

The categories non-renewable energy, global warming potential, eutrophication, terrestrial ecotoxicity, terrestrial biodiversity reduction and aquatic biodiversity loss were compared with existing studies in Chapters 4.4.6 and 5.6.1 in order to verify plausibility. The values obtained in the present study for the categories ozone formation, acidification, aquatic ecotoxicity and human toxicity are further assessed for plausibility here, by comparing them with the values obtained in a previous study on crop rotation systems in Switzerland [131] (henceforthe referred to as the "reference study"). Thus only land competition could not be assessed for plausibility due to the lack of a value for comparison. Ozone formation and acidification potentials were re-calculated for the regional scenarios assessed here, using

the same method [127] as in the reference study in order to allow comparison. The values for the reference study, and for the regional scenarios assessed, here are given in Table A4.2:

Table A4.2: *comparison of the order of magnitude of some impacts between a reference study and the regional scenarios assessed here.*

Impact category	Unit	Reference study	Regional scenarios
Ozone formation	kg C_2H_4 eq./ha*y	0.64 – 1	1.08 – 1.15
Acidification	Kg SO_2 eq./ha*y	8 – 88	77 – 93
Aquatic ecotoxicity	1,4-DB eq./ha*y	609 – 6598	552 – 1123
Human toxicity	1,4-DB eq./ha*y	329 – 1417	1226 – 1619

Table A4.2 shows that the outcomes of the regional scenarios are in the same order of magnitude as those of the reference study, and are thus considered plausible, whereby ozone formation, acidification and human toxicity are rather high whereas aquatic ecotoxicity is rather low.

A4.3. Yields

Figure A4.1 provides the yields of the regional scenarios assessed here.

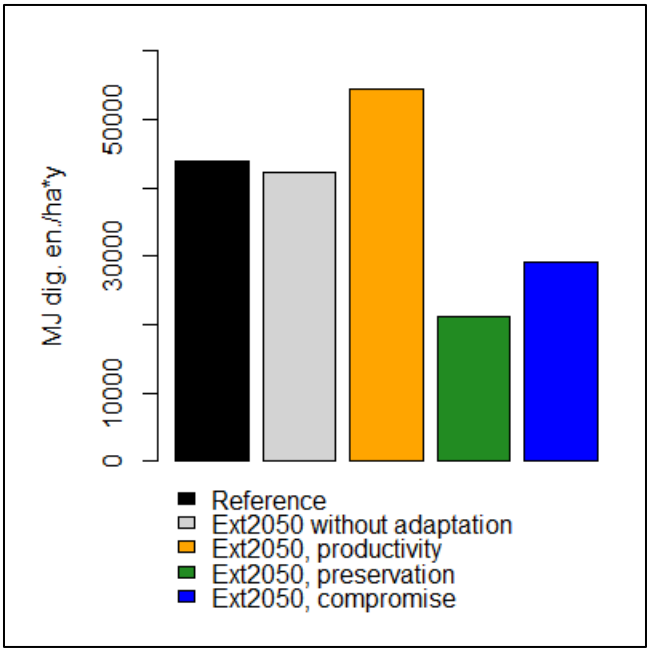


Figure A4.1: *on-field yields (expressed as MJ dig. en. for humans, including both crop and animal production[3]) for the regional scenarios assessed here.*

[3] NB: as explained in the description of the regional optimization model in Annex A4.1, the number of livestock did not vary between scenarios; only the crop production amount thus varies between the scenarios.

A4.4. Correlation matrix

Table A4.3 provides the correlations between impact categories, for all the indicators and scenarios assessed in the LCA of regional adaptation to climate change.

Table A4.3: *correlation between impacts (initial indicator selection) for the regional scenarios assessed here, with functional unit MJ dig. en.*

	Non-renewable energy	GWP	Ozone formation	Acidification	Terrestrial Eutrophication	Marine eutrophication	Freshwater eutrophication	Human toxicity	Terrestrial Ecotoxicity	Aquatic ecotoxicity	Land competition	Aquatic biodiversity loss	Terrestrial biodiversity reduction
Non-renewable energy	1.00												
GWP	0.98	1.00											
Ozone formation	0.99	0.99	1.00										
Acidification	0.95	0.96	0.98	1.00									
Terrestrial Eutrophication	0.95	0.96	0.98	1.00	1.00								
Marine eutrophication	0.92	0.89	0.90	0.89	0.88	1.00							
Freshwater eutrophication	-0.84	-0.86	-0.86	-0.86	-0.85	-0.59	1.00						
Human toxicity	0.99	0.98	1.00	0.98	0.98	0.92	-0.85	1.00					
Terrestrial Ecotoxicity	0.95	0.98	0.99	0.99	0.99	0.85	-0.89	0.98	1.00				
Aquatic ecotoxicity	0.81	0.86	0.86	0.85	0.85	0.88	-0.53	0.83	0.84	1.00			
Land competition	0.94	0.92	0.96	0.98	0.98	0.90	-0.82	0.97	0.96	0.79	1.00		
Aquatic biodiversity loss	-0.61	-0.74	-0.69	-0.64	-0.64	-0.44	0.67	-0.61	-0.71	-0.72	-0.49	1.00	
Terrestrial biodiversity reduction	0.80	0.68	0.69	0.62	0.60	0.77	-0.54	0.74	0.59	0.41	0.69	-0.10	1.00

A4.5. Inventories of regional scale scenarios

Table A4.4 provides the inventories of all regional scale scenarios for the entire Broye region and the entire 5-year crop rotation period.

***Table A4.4**: inventories of all regional scale scenarios (Broye region, 5 year period), according to ecoinvent and SALCA nomenclature (continued pages 186 to 188).*

		Scenario				
Flow	Unit	Reference	Ext2050 without adaptation	Ext2050, productivity	Ext2050, preservation	Ext2050, compromise
Resources						
Carbon dioxide, in air	kg	-2.7E+09	-2.43E+09	-3.83E+09	-2.19E+09	-2399500000
Energy, gross calorific value, in biomass, biotic	MJ	-3.3E+10	-2.94E+10	-4.65E+10	-2.654E+10	-2.9085E+10
Water, river	l	5319200	7952400	231210000	25000	6043700
Water, well, in ground	l	6056400	6056400	20228000	20219000	6192100
Materials/fuels						
[sulfonyl]urea-compounds, at regional storehouse/CH S	kg	4831.4	4831.4	3326.2	2774.3	4620.6
[thio]carbamate-compounds, at regional storehouse/CH S	kg	15626	15626	16357	10912	13066
2,4-D, at regional storehouse/CH S	kg	6319.9	6319.9	7375.5	5185.4	5730.5
Acetamide-anillide-compounds, at regional storehouse/CH S	kg	23245	23245	24977	21186	22609
Agricultural machinery, general, production/CH/I S	kg	3382200	3382200	2381700	2566700	3133700
Agricultural machinery, tillage, production/CH/I S	kg	1407700	1407700	1412200	1052400	1244100
Ammonium nitrate, as N, at regional storehouse/RER S	kg	10411000	10411000	10050000	4654900	10638000
Ammonium sulphate, as N, at regional storehouse/RER S	kg	800860	800860	773060	343420	818260
Atrazine, at regional storehouse/CH S	kg	5355.3	5355.3	9329.5	5775.5	3052.3
Barley seed IP, at regional storehouse/CH S	kg	1432800	1432800	3979200	1578500	4845500
Benzimidazole-compounds, at regional storehouse/CH S	kg	23767	23767	25115	14907	19621
Benzo[thia]diazole-compounds, at regional storehouse/CH S	kg	19436	19436	22541	14000	15425
Benzoic-compounds, at regional storehouse/CH S	kg	5534.8	5534.8	5859.3	3626.6	5250.5
Bucket milking system, at farm/p/CH/I U	p	1030.5	1030.5	1923.5	2070.1	1660.2
Calcium ammonium nitrate, as N, at regional storehouse/RER S	kg	5205600	5205600	5024900	2232200	5318700
Cast iron, at plant/RER S	kg	90250	90250	66750	173500	98500
Combine harvesting/CH S	ha	110580	110580	108250	70025	111020
Concentrate feed IP, average composition CH, at consumer/kg/CH U	kg	1.07E+08	107353654	107353654	107353654	107353654
Copper oxide, at plant/RER S	kg	178060	178060	3645.2	5327.6	68418
Cubicle housing, cattle, wood construction, non-insulated, at farm/p/CH/I U	p	179.34	179.34	334.77	360.27	288.94
Cyclic N-compounds, at regional storehouse/CH S	kg	32429	32429	36977	20992	29885
Deep-litter housing, wood construction, non-insulated, at farm/m2/CH/I U	m2	6643.4	6643.4	12401	13346	10703
Diammonium phosphate, as P2O5, at regional storehouse/RER S	kg	2943700	2943700	3177900	2061900	3137800
Dicamba, at regional storehouse/CH S	kg	587.39	587.39	621.84	384.88	557.22
Diesel combustion, in tractor/kg/CH U	kg	21492000	21492000	18957000	17151000	19959000
Diesel, at regional storage/CH S	kg	21492000	21492000	18957000	17151000	19959000
Dinitroaniline-compounds, at regional storehouse/CH S	kg	46362	46362	48449	36554	40436
Diphenylether-compounds, at regional storehouse/CH S	kg	2335.1	2335.1	1657.5	1257.7	2136.1
Dung slab/CH/I S	m2	14581	14581	2602.1	10032	11378
Exercise yard, floored, perforated, at farm/m2/CH/I U	m2	5613.5	5613.5	10478	11277	9043.9
Feed-concentrate silo, steel, at farm/m3/CH/I U	m3	2921.3	2921.3	4089.8	4927.2	2823.9
Field-cured hay, perm. meadow, IP, int, plain reg, at farm/kg/CH U	kg		22902.462	3497011.5		
Fully slatted floor, cattle, wood construction, non-insulated, at farm/p/CH/I U	p	46.133	46.133	86.115	92.676	74.325
Garage, wood, non-insulated, fire-protected, at farm/m2/CH/I U	m2	6582.1	6582.1	5507.9	5941	6408.4
Glyphosate, at regional storehouse/CH S	kg	1046.9	1046.9	641.76	1666.6	1486.6
Grass silage, sil. bales, perm. meadow, IP, int, plain reg, at farm/kg/CH U	kg		1006878.1	153741738		
Grass, perm. meadow, IP, int, plain reg, at farm/kg/CH U	kg		645860.43	98617404		
Harvester, production/CH/I S	kg	71188	71188	43640	113330	101090
Harvesting, by complete harvester, beets/CH S	ha	18000	18000	35900	7375	2475
Hay intensive IP, at farm/CH U	kg		318975.51	48704853		
Linuron, at regional storehouse/CH S	kg	2426.6	2426.6	1360.9	862.37	1645.2
Magnesium oxide, at plant/RER S	kg	3359100	3359100	3732800	2081700	4095500
Maize seed IP, at regional storehouse/CH S	kg	1057000	1057000	1861700	1174000	602820
MCPA, at regional storehouse/CH S	kg	9784.4	9784.4	10080	6995.3	9575.9
Milk cooling tank, at farm/m3/CH/I U	m3	70.092	70.092	130.84	140.81	112.93
Milk powder IP/CH U	kg	1858681	1858681	1858681	1858681	1858681
Milking parlour, at farm/p/CH/I U	p	29.89	29.89	55.795	60.046	48.156
Nitrile-compounds, at regional storehouse/CH S	kg	12346	12346	14234	9120.2	13158
Organophosphorus-compounds, at regional storehouse/CH S	kg	3699.9	3699.9	2176.6	2579.4	4552.7
Passenger car/RER/I S	p	297.29	297.29	331.52	295.27	287.39
Pesticide unspecified, at regional storehouse/CH S	kg	5681.4	5681.4	6750.9	4251.2	5249.5
Petrol combustion, unleaded, in motor mower/kg/CH U	kg	2909000	2909000	2878500	2736300	2734700
Petrol, unleaded, at regional storage/CH S	kg	2909000	2909000	2878500	2736300	2734700
Phenoxy-compounds, at regional storehouse/CH S	kg	8761.9	8761.9	6919.6	6641.8	8404.7
Phosphate rock, as P2O5, beneficiated, dry, at plant/MA S	kg	2370000	2370000	2538300	1546800	2524300

Table A4.4 continued

Flow	Unit	Scenario				
		Reference	Ext2050 without adaptation	Ext2050, productivity	Ext2050, preservation	Ext2050, compromise
Pipeline milking system, at farm/p/CH/I U	p	1030.5	1030.5	1923.5	2070.1	1660.2
Potassium chloride, as K2O, at regional storehouse/RER S	kg	12574000	12574000	10348000	7318000	15412000
Potassium sulphate, as K2O, at regional storehouse/RER S	kg	802620	802620	660510	467100	983720
Potato haulm cutting/CH S	ha	15875	15875	325	475	6100
Potato seed IP, at regional storehouse/CH S	kg	40005000	40005000	819000	1197000	15372000
Pyretroid-compounds, at regional storehouse/CH S	kg	339.91	339.91	228.65	231.11	395.96
Pyridazine-compounds, at regional storehouse/CH S	kg	3677.1	3677.1	7333.8	1506.6	505.61
Rape seed IP, at regional storehouse/CH S	kg	100100	100100	2062.5	81400	167340
Round wood, softwood, debarked, u=70% at forest road/RER S	m3	315880	315880	233620	607250	344750
Shed, large, wood, non-insulated, fire-unprotected, at farm/m2/CH/I U	m2	24973	24973	19875	19365	22947
Silage maize, IP, plain region, at farm/kg/CH U	kg	16357.64	2428983.2			1050739.61
Single superphosphate, as P2O5, at regional storehouse/RER S	kg	197500	197500	211520	128900	210350
Slurry store and processing, operation/CH S	m3	3041300	3041300	542730	2092400	2373300
Slurry tanker, production/CH/I S	kg	70337	70337	131290	141300	113320
Sowing/CH S	ha	59100	59100	107500	51700	25900
Soya bean meal, at oil mill/kg/CH U	kg	21131875	21131875	21131875	21131875	21131875
Sugar beet seed IP, at regional storehouse/CH S	kg	37800	37800	75390	15488	5197.5
Tethered housing, cattle, concrete wall, at farm/p/CH/I U	p	712.6	712.6	1330.2	1431.5	1148.1
Thomas meal, as P2O5, at regional storehouse/RER S	kg	493740	493740	528810	322240	525890
Tractor, production/CH/I S	kg	2393000	2393000	1989100	1850900	2160300
Trailer, production/CH/I S	kg	1351600	1351600	1184300	1082000	1264400
Transport, barge/RER S	tkm	65523000	65523000	63855000	30934000	67784000
Transport, freight, rail/CH S	tkm	10892000	10892000	10483000	5647800	11762000
Transport, lorry 16-32t, EURO3/RER S	tkm	10892000	10892000	10483000	5647800	11762000
Transport, tractor and trailer/CH S	tkm	12960000	12960000	25848000	5310000	1782000
Transport, van <3.5t/CH S	tkm	795240	795240	206220	145780	381890
Triazine-compounds, at regional storehouse/CH S	kg	7884.6	7884.6	13834	4438.1	2251.5
Triple superphosphate, as P2O5, at regional storehouse/RER S	kg	4048700	4048700	4336200	2642400	4312300
Urea, as N, at regional storehouse/RER S	kg	3603900	3603900	3478800	1545400	3682200
Wheat seed IP, at regional storehouse/CH S	kg	9544500	9544500	6471000	5310000	3910500
Zinc coating, coils/RER S	m2	22562	22562	16688	43375	24625
Emissions to air (low population)						
Ammonia	kg	10446000	10446000	6676100	10171000	9217500
Carbon dioxide, fossil	kg	5658100	5658100	5461700	2426300	5781000
Dinitrogen monoxide	kg	1704100	1738200	1204100	1313100	1481500
Methane, biogenic	kg	2054600	2054600	367820	1414000	1603600
Nitrogen oxides	kg	357850	365030	252850	275750	311120
Emissions to water						
Cadmium, ion, groundwater	kg	9.4755	9.5871	9.372	7.6251	8.8728
Cadmium, ion, river	kg	225.34	284.85	368.02	72.862	116.31
Chromium, ion, groundwater	kg	4322.3	4325.4	4304.4	3862.8	4149.9
Chromium, ion, river	kg	25399	32068	41376	8617	13159
Copper, ion, groundwater	kg	1018.5	1121	1270.4	740.2	812.73
Copper, ion, river	kg	18487	23254	28971	6412.8	9482.8
Lead, groundwater	kg	41.397	42.452	31.858	40.225	36.509
Lead, river	kg	5117.2	6271.1	6408.9	1857.2	2349
Mercury, groundwater	kg	52.343	66.058	48.869	11.474	18.14
Mercury, river	kg	10.712	12.253	6.9185	7.012	4.1487
Nickel, ion, groundwater	kg	35.598	44.611	19.918	4.3293	11.406
Nickel, ion, river	kg	17019	21358	26375	5818.5	8686.9
Nitrate, groundwater	kg	30187000	43028000	53121000	20536000	30718000
Phosphate, groundwater	kg	37531	37531	39083	26256	36489
Phosphate, river	kg	21417	21417	14011	18092	24782
Phosphorus, river	kg	915910	1175300	1636600	286720	501960
Zinc, ion, groundwater	kg	4376.5	4549.2	4226.8	4137.2	3898.7
Zinc, ion, river	kg	26664	33160	38938	9883.3	13419
Emissions to soil (agricultural)						
2,4-D	kg	6319.9	6319.9	7375.5	5185.4	5730.5
Acetamiprid	kg	34.517	34.517	0.7112	28.069	57.702
Acibenzolar-s-methyl	kg	144.61	144.61	98.045	80.454	59.25
Aclonifen	kg	1298.9	1298.9	26.591	38.864	499.09
Alachlor	kg	3452.4	3452.4	6014.4	3723.3	1967.7

Table A4.4 continued

Flow	Unit	Scenario				
		Reference	Ext2050 without adaptation	Ext2050, productivity	Ext2050, preservation	Ext2050, compromise
Alpha-cypermethrin	kg	36.547	36.547	24.084	24.95	42.961
Amidosulfuron	kg	124.98	124.98	118.7	110.49	132.95
Asulam	kg	2791.7	2791.7	1711.4	4444.1	3964.1
Atrazine	kg	5355.3	5355.3	9329.5	5775.5	3052.3
Azoxystrobin	kg	2223.5	2223.5	2633.4	1529.5	2505.1
Bentazone	kg	19292	19292	22443	13919	15366
Bifenthrin	kg	29.683	29.683	19.56	20.264	34.892
Bromoxynil-ester	kg	2770.8	2770.8	3523.2	2180.8	2301.7
Cadmium	kg	656.52	596.9	507.36	562.43	791.42
Carbetamide	kg	1654.5	1654.5	34.091	1345.4	2765.9
Carbofuran	kg	600	600	1196.7	245.83	82.5
Carfentrazone-ethyl	kg	147.2	147.2	155.83	96.451	139.64
Chloridazon	kg	3677.1	3677.1	7333.8	1506.6	505.61
Chlorothalonil	kg	22428	22428	22444	14358	19437
Chlorthal	kg	5534.8	5534.8	5859.3	3626.6	5250.5
Chromium	kg	-13418	-20091	-30007	-458.54	-633.26
Clethodim	kg	556.87	556.87	376.89	242.49	423.91
Clomazone	kg	316.18	316.18	6.5147	257.12	528.56
Clopyralid	kg	148.04	148.04	247.05	101.91	65.109
Copper	kg	193030	188160	-10719	23444	83326
Cycloxydim	kg	855.03	855.03	852.78	601.79	667.24
Cypermethrin	kg	204.07	204.07	134.48	139.32	239.88
Cyproconazole	kg	1016.1	1016.1	1203.5	698.99	1144.8
Cyprodinil	kg	3811.7	3811.7	4514.4	2622	4294.4
Deltamethrin	kg	58.772	58.772	44.379	38.969	64.696
Desmedipham	kg	252.77	252.77	504.14	103.57	34.756
Diazinon	kg	1298.6	1298.6	855.77	886.56	1526.5
Dicamba	kg	587.39	587.39	621.84	384.88	557.22
Dichlobenil	kg	392.58	392.58	240.66	624.96	557.46
Difenoconazole	kg	1807.7	1807.7	1225.6	1005.7	740.62
Diflufenican	kg	638.86	638.86	676.32	418.6	606.04
Dimefuron	kg	827.27	827.27	17.045	672.72	1382.9
Dimethenamid	kg	6078.7	6078.7	10835	5905.6	3044.2
Dimethoate	kg	240	240	478.67	98.333	33
Epoxiconazole	kg	5652	5652	6811.2	3918.4	6533.6
Ethofumesate	kg	1338.9	1338.9	2670.4	548.59	184.1
Famoxadone	kg	1905.8	1905.8	2257.2	1311	2147.2
Fenpropimorph	kg	4523.6	4523.6	5479.6	3143.4	5269.2
Fluazifop	kg	1025.2	1025.2	592.09	505.03	921.66
Fluroxypyr	kg	1116.5	1116.5	1182	731.59	1059.2
Flusilazole	kg	2035.4	2035.4	2410.7	1400.2	2293.2
Glyphosate	kg	1046.9	1046.9	641.76	1666.6	1486.6
Haloxyfop-P-methyl	kg	196.84	196.84	113.68	96.966	176.96
Heptenophos	kg	194.1	194.1	387.12	79.527	26.689
Hexaconazole	kg	903.83	903.83	612.78	502.84	370.31
Ioxynil	kg	1001	1001	1059.7	655.87	949.56
Isoxadifen-ethyl	kg	12600	12600	13339	8256.2	11953
Kresoxim-methyl	kg	1588.2	1588.2	1881	1092.5	1789.3
Lambda-cyhalothrin	kg	29.396	29.396	18.37	20.274	35.334
Lead	kg	-1751.2	-2914.1	-4157.5	1366.3	659.46
Lenacil	kg	2468.6	2468.6	4923.4	1011.4	339.43
Linuron	kg	2426.6	2426.6	1360.9	862.37	1645.2
MCPA	kg	9784.4	9784.4	10080	6995.3	9575.9
MCPB	kg	2490.1	2490.1	671.3	1709.7	2041.1
Mecoprop-P	kg	5155.3	5155.3	5066.3	4200.4	5304.4
Mercury	kg	361.36	358.76	215.66	378.66	312.1
Metamitron	kg	5040.5	5040.5	10053	2065.2	693.07
Metazachlor	kg	3944.3	3944.3	81.271	3207.5	6593.8
Methidathion	kg	210	210	418.83	86.041	28.875
Methomyl	kg	300	300	598.33	122.92	41.25
Metolachlor, (S)	kg	4390.7	4390.7	7935.5	3975.8	2011.4
Metoxuron	kg	2826.3	2826.3	2992	1851.9	2681.1

Table A4.4 continued

Flow	Unit	Scenario				
		Reference	Ext2050 without adaptation	Ext2050, productivity	Ext2050, preservation	Ext2050, compromise
Metribuzin	kg	599.42	599.42	12.272	17.936	230.33
Metsulfuron-methyl	kg	25.319	25.319	25.306	16.201	21.899
Monolinuron	kg	950.09	950.09	19.451	28.428	365.07
Napropamide	kg	4861.9	4861.9	100.18	3953.6	8127.6
Nickel	kg	-11415	-15768	-22366	-731.35	-3590.9
Nicosulfuron	kg	102.75	102.75	179	110.81	58.562
Orbencarb	kg	1739.2	1739.2	35.605	52.038	668.28
Pendimethalin	kg	39855	39855	45381	32726	33273
Phenmedipham	kg	3939.5	3939.5	7857.2	1614.1	541.68
Phosalone	kg	1757.2	1757.2	36.207	1429	2937.6
Pirimicarb	kg	1391.4	1391.4	916.89	949.89	1635.6
Pronamide	kg	517.04	517.04	10.653	420.45	864.34
Propaquizafop	kg	318.96	318.96	184.21	157.12	286.74
Propiconazole	kg	602.55	602.55	408.52	335.23	246.87
Pyraclostrobin (prop)	kg	2957.2	2957.2	3502.4	2034.2	3331.7
Quizalofop ethyl ester	kg	136.7	136.7	78.945	67.337	122.89
Simazine	kg	2055	2055	3580	2216.2	1171.2
Spinosad	kg	120.5	120.5	2.4828	97.986	201.43
Sulcotrione	kg	1233	1233	2148	1329.7	702.75
Tepraloxydim	kg	136.7	136.7	78.945	67.337	122.89
Thiacloprid	kg	240.99	240.99	4.9655	195.97	402.87
Thifensulfuron-methyl	kg	164.32	164.32	163.65	122.55	134.96
Triazamate	kg	33.6	33.6	67.013	13.767	4.62
Triclopyr	kg	139.58	139.58	85.568	222.21	198.21
Trifloxystrobin	kg	1191.1	1191.1	1410.8	819.38	1342
Trifluralin	kg	5208.8	5208.8	3041.1	3789.3	6664
Zinc	kg	78158	71490	26683	93221	77354
Waste to treatment						
Disposal, wood untreated, 20% water, to municipal incineration/CH S	kg	412.91	412.91	305.39	793.79	450.65

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