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Evaluation environnementale des systèmes agricoles urbains en Afrique de l'Ouest: Implications de la diversité des pratiques et de la variabilité des émissions d'azote dans l'Analyse du Cycle de Vie de la tomate au Bénin

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ELSA
Environmental Lifecycle & Sustainability Assessment

UR Fonctionnement agro ecologique et performances des systèmes de culture horticoles

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Table des matières

Liste des figures **1**

Liste des tableaux **3**

Liste des abréviations **5**

Introduction générale **7**

Nourrir la planète : un enjeu global et des solutions locales 7
Les méthodes d'évaluation de l'agriculture 8
ACV et qualité des données d'inventaire 9
Evaluation environnementale de la tomate produite en agriculture urbaine au Bénin 12
Contexte et objectifs de la thèse 14
Plan de la thèse et démarche scientifique 15
Références bibliographiques 17

Chapitre 1. Analyse du cycle de vie des systèmes de culture maraichers: une revue centrée sur la diversité des systèmes de culture et l'estimation des émissions au champ 21

Avant-propos **22**

Résumé **23**

Life cycle assessment of vegetable products: a review focusing on cropping systems diversity and the estimation of field emissions **25**

Abstract **26**

1.2. Materials and methods 28
1.2.2. Typology of cropping systems and aggregation of results 28

1.3. Presentation of LCA studies selected 29
1.3.1. Goal and scope 29
1.3.2. Data and methods for LCI 31
1.3.3. Input flows and yields 34
1.3.4. LCIA results 37

1.4. Relevance of the typology to compare LCIA results from a diversity of cropping systems and vegetable products 41
1.4.1. Inherent variability of cropping systems and representativeness issues 42
1.4.2. Discrepancies in methodological choices 42
1.4.3. Weaknesses in reporting, a limitation to our typology approach 44
1.4.4. Relevance and validity of the methods used for the estimation of reactive nitrogen emissions 45

1.5. Conclusions and recommandations 48
1.6. References 52

Contribution du chapitre 1 et perspectives dans le cadre de la thèse **55**

Chapitre 2. Méthodes et données pour une analyse du cycle de vie fiable et représentative dans un contexte de faible disponibilité en données 57

Avant-propos.....58

Résumé 59

Life Cycle Inventory for tomato urban gardens in Benin: low performances and high environmental risks systems 61

Abstract.....61

2.1. Introduction.....62
2.2. Materials and methods.....63
2.2.1. Study sites and characterization of cropping systems 63
2.2.2. Sampling cropping systems for monitoring.....64
2.2.3. Cropping systems data for environmental inventories 65
2.2.4. Comparison with scenarios based on the literature.....66
2.2.5. Assessment of the quality of LCI data 67

2.3. Results and discussion 68
2.3.1. Diversity of cropping systems sampled.....68
2.3.2. Crop protection: frequent applications of pesticides at excessive rates 71
2.3.3. Intensive fertilization management: important N and P surplus.....73
2.3.4. Variable and low tomato yields 77
2.3.5. Prospects for the Life cycle assessment of out-of season tomato in Benin 79

2.4. Conclusions.....80
2.5. References.....81

Contribution du chapitre 2 et perspectives dans le cadre de la thèse 83

Chapitre 3. Boucler le bilan d'azote à l'échelle du système de culture : perspectives pour l'estimation des émissions d'azote réactif pour la tomate en maraîchage urbain au Bénin 85

Avant-Propos 86

Résumé 87

Closing the nitrogen budget for tomato cropping systems in African urban gardens 89

3.1. Introduction.....89
3.2. Materials and methods.....91
3.2.1. An approach combining a crop model and a nitrogen budget 91
3.2.2. Calculation of Nr emissions in STICS 93
3.2.3. Parameterization of the model for the case study 96
3.2.4. Statistical analysis.....99

3.3. Results and discussion 100
3.3.1. Quality of prediction and model soundness for yields 100
3.3.2. Analysis of drivers for nitrogen losses as simulated by STICS 102
3.3.3. Evaluation of the combined approach 106

3.4. Conclusion 112
3.5. Reference 113

Contribution du chapitre 3 et perspectives dans le cadre de la thèse 117

Chapitre 4. Exploration de la variabilité des impacts environnementaux calculés par l'analyse du cycle de vie de la tomate en maraîchage urbain au Bénin 119

Avant-Propos 120

Résumé 121

Investigation of the variability of life-cycle assessment of tomato production from Beninese urban garden.....123

4.1. Introduction.....123
4.2. Material and method.....124
4.2.1. Goal and scope definition.....124
4.2.2. Inventory data.....127
4.2.3. Design of and comparison with literature LCIA reference scenarios 132
4.2.4. Design and analysis of simulated cropping system scenarios with the crop model STICS .133

4.3. Results 135
4.3.1. LCIA results across the modalities and the weighted mean 135

4.4. Discussion.....148
4.4.1. Impact of the irrigation management 148
4.4.2. Impact of fertilization management 148
4.4.3. Impact of crop protection management.....149

4.5. Conclusions.....150
4.6. References.....151

Chapitre 5. Discussion générale.....155

5.1. Introduction.....156
5.2. Une approche de type méta-analyse pour capitaliser sur le matériau bibliographique.....159
5.3. Contribution à la réalisation d'inventaires environnementaux.....161
5.3.1. Proposition d'un protocole de collecte des données pour réaliser des inventaires représentatifs de la diversité des systèmes de production 161
5.3.2. Proposition d'une approche intégrée pour l'analyse de la variabilité des émissions 163

5.4. Contribution à l'évaluation environnementale de l'agriculture urbaine par la méthode ACV .165
5.4.1. Impact environnemental de la tomate produite en maraîchage urbain au Bénin 165
5.4.2. Résultats de l'analyse de scénario : vers un raffinement des méthodes d'estimation au champ 166
5.4.3. Identification des "hot-spots" à l'échelle de la population 167

5.5. Recommandations et perspectives pour améliorer la robustesse des résultats d'ACV en contexte tropical 169
5.5.1. Intégrer les aspects socio-économiques.....169
5.5.2. Collecter des données spécifiques sur les flux à la parcelle 170
5.5.3. Mettre à disposition des méthodes et des outils adaptés.....170
5.5.4. Améliorer la prise en compte des pesticides, des engrais organiques et de la consommation en eau.....171

5.6. Conclusions.....173
5.7. Références bibliographiques 174

ANNEXES 177

Annex 1. Global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP) expressed per square meter for a vegetable crop cycle from cradle-to-harvest for the 72 systems aggregated per cropping system type, product group and reviewed paper**. 179**

Annexe 2. Liste des producteurs de tomate de contre saison pour la saison 2011-2012 sur les communes de Cotonou, Ouidah et Grand-Popo**.....180**

Annexe 3. Protocole d'enquête pour la collecte des données qualitatives par les conseillers auprès des 12 agriculteurs sélectionnés **.....184**

Annexe 4. Liste des producteurs de tomate de contre saison pour la saison 2011-2012 sur les communes de Cotonou, Ouidah et Grand-Popo**.....188**

Annexe 5. Exemple des données complètes collectées pour l'itinéraire technique pour la parcelle 6**.....194**

Annexe 6. Résultats des analyses de sol pour les 12 parcelles échantillonnées **.....196**

Annexe 7. Résumé des données climatiques produites par la station météo du centre de recherche AfricaRice localisé à Cotonou**.....197**

Annex 8. LCIA results from cradle-to-harvest per hectare of out-of-season tomato in Benin for the 10 modalities accounting for 4 criteria, and the weighted mean**.....199**

Portefeuille de Compétences.....199

Liste des figures

Introduction

Figure 1 Relations entre les différentes étapes du processus d'analyse du cycle de vie (ISO, 2006)....**10**
Figure 2 Plan de la thèse et démarche scientifique **15**

Chapitre 1

Figure 1.1 Global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP) expressed per kg of fresh yield from cradle-to-harvest for the 72 systems aggregated per cropping system type, product group and reviewed paper.....**40**

Figure 1.2 Résultats du chapitre 1 (boite verte) et contributions pour les chapitres suivants (rouge)...**55**

Chapitre 2

Figure 2.1 Densité de population au Bénin (Center for International Earth Science Information Network) **58**
Figure 2.2 The typology-based procedure to (1) sample 12 diverse cropping systems and (2) create a representative inventory for out-of-season tomato cropping systems in studied urban areas in Benin (2011-2012 - Cotonou, Ouidah, Grand Popo).. .....**69**
Figure 2.3 Treatment frequency index (TFI) per targeted pest: insect, disease and nematod for each active substance used for crop protection, for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean). **72**
Figure 2.4 Nitrogen and phosphorus inputs for main nutrient horizontal fluxes at field scale; for cropping system aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean) **74**
Figure 2.5 Nitrogen and phosphorus outputs for main nutrient fluxes at field scale for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean) **75**
Figure 2.6 Nitrogen and phosphorus surplus at field scale for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean).....**77**

Figure 2.7 Résultats du chapitre 2 (boite violette) et contribution pour les chapitres suivants (rouge) **83**

Chapitre 3

Figure 3.1 Conceptual model for output variables of interest (circles) as simulated in STICS: main influencing processes (white rectangle) and driving parameters (grey rectangles) **92**
Figure 3.2 Effect of soil pH on volatilization from mineral fertilizers **94**
Figure 3.3 Effect of dry matter content of fertilizer and soil tillage on the volatilizable fraction **94**
Figure 3.4 Effects of pH (FPHN), Soil relative water content (FHN) and soil temperature (FTN) on nitrification **95**
Figure 3.5 Effects of soil temperature (FDENT), soil nitrate content (FDENNO3) and soil properties (FDENW) on denitrification **96**
Figure 3.6 Summary of climate variables for the 2011/2012 dry season at Cotonou, Benin, West Africa **97**
Figure 3.7 Simulated versus observed yields **100**
Figure 3.8 Yield residues versus observed yields **100**

Figure 3.9 Simulated and observed yields versus cumulative global radiation over the crop cycle ...**101**
Figure 3.10 Simulated and observed yields versus cumulative temperature over the crop cycle **101**
Figure 3.11 Simulated and observed yields versus total water **101**
Figure 3.12 Simulated and observed yields versus nitrogen inputs **101**
Figure 3.13 Simulated and observed yields versus plant density **101**

Figure 3.14 Total Nr emissions versus nitrogen input from mineralization of soil organic matter, organic and mineral fertilizers **102**
Figure 3.15 NH_4 emissions factors versus soil pH **103**
Figure 3.16 NH_4 emissions factors for mineral fertilizers versus total water **103**
Figure 3.17 N_2O emission factors versus soil moisture at maximal capacity for all cropping systems **104**
Figure 3.19 N_2O emission factors for nitrification versus soil pH **104**
Figure 3.18 N_2O emission factors versus soil moisture at maximal capacity for sandy soils **104**
Figure 3.20 NO_3 emissions versus the total nitrified nitrogen **105**
Figure 3.21 NO_3 emissions versus total volume of water **105**
Figure 3.22 Nitrogen budget simulated with STICS for the 12 cropping systems **106**
Figure 3.23 Nitrogen input from soil mineralization versus organic fertilizers **107**
Figure 3.24 Simulated harvested nitrogen versus simulated yields **107**

Figure 3.25 Résultats du chapitre 3 (boite violette) et contribution pour les chapitres suivants (rouge) **117**

Chapitre 4

Figure 4.1 Les contributions des chapitres précédents (rouge) pour répondre à l'objectif 4 ..... **120**

Figure 4.2 LCIA results from cradle-to-harvest per kilogram of out-of-season tomato in Benin of the 10 modalities accounting for 4 criteria, and the weighted mean for 5 impacts categories: water deprivation potential (WDP), climate change potential (CCP), human toxicity potential (HTP), photochemical ozone formation potential (POFP) and marine ecotoxicity potential (METP) **137**
Figure 4.3 LCIA results from cradle-to-harvest per kilogram of out-of-season tomato in Benin of the 10 modalities accounting for 4 criteria, and the weighted mean; for 3 impacts categories: terrestrial acidification potential (TAP), freshwater and marine eutrophication potential (FEP and MEP) **139**
Figure 4.4 LCIA results from cradle-to-harvest per kilogram of out-of-season tomato in Benin of the 10 modalities accounting for 4 criteria, and the weighted mean; for 2 impacts categories: terrestrial and freshwater ecotoxicity potential (TETP and FETP) **141**
Figure 4.5 Scenario analysis for climate change potential (CCP), terrestrial acidification potential (TAP) and marine eutrophication potential (MEP) to minimum (Min) and maximum (Max) scenarios for 4 drivers influencing N_r emissions: nitrogen rate, water volume, soil pH and maximum soil moisture **147**

Chapitre 5

Figure 5.1 Principaux résultats de la thèse et progression de la discussion générale **158**

Liste des tableaux

Chapitre 1

Table 1.1 LCI methods to estimate reactive nitrogen field emissions: guidelines and references, calculation method using emission factors and parameters, validity domain and uses in selected studies **33**
Table 1.2 Inputs flows for the field production stage expressed per ton of fresh yield and fresh yield for the 72 systems aggregated by cropping system type, product group and reference. **36**
Table 1.3 Summary of recommendations related to key parameters to be included in LCI and methods to be used for the estimation of reactive nitrogen (N_r) emissions to account for specificities of vegetable cropping systems **49**

Chapitre 2

Table 2.1 Distribution of the 10 types at the whole population scale (typology based on the irrigation system, the water resource and the planting date) in terms of area (m²) and number of farms for each district **70**
Table 2.2 Average number of applications and associated coefficient of variation per targeted pest: insect, disease and nematod for cropping systems aggregated by modality for each criterion: irrigation system, water resource and planting date and for the scenario representative for the whole population (WM). **71**
Table 2.3 Average and range of variation for emission factors for the scenario representative (weighted mean) where emissions were estimated based on the literature and for the Rotation scenario where emissions were measured on field (18-month period) **76**
Table 2.4 Cropping system data over one tomato crop cycle for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (WM). **78**
Table 2.5 Data quality indicators (Weidema and Wesnaes, 1996) for our LCI data set for out-of-season tomato in Benin. The best score is 1, the worst is 5 ..... **80**

Chapitre 3

Tableau 3.1 Facteurs influençant la variation des émissions azotées au champ pour les cultures maraichères **86**
Table 3.2 Site-specific soil data used to feed the STICS model from 12 tomato cropping systems in Benin **97**
Table 3.3 Management data used to feed the STICS model from 12 tomato cropping systems in Benin **98**
Table 3.4 General parameters used to adapt the STICS crop model to tropical conditions **99**
Table 3.5 Performances of the structured estimation methods for Nr emissions **111**

Chapitre 4

Table 4.1 Weighting factors based on area distribution used to calculate the weighted mean and subsequent contribution of modalities within each criterion **126**
Table 4.2 Parameters used for the inventory modeling of production processes per modality of each criterion and for the weighted mean (WM) **129**
Table 4. 3 Agricultural input data per modality of each criterion and for the weighted mean (WM). **131**
Table 4.4 Characteristics for literature vegetable scenarios and Beninese out-of-season tomato **132**
Table 4.5 Key influencing drivers tested and corresponding favorable and unfavorable scenarios to emissions for the scenario analysis of LCIA results **134**
Table 4.6 LCIA results per kg of product for WDP, global warming, acidification and eutrophication potentials for literature vegetable scenarios and Beninese out-of-season tomato (this study) **143**
Table 4.7 N_r emission factors obtained with the Default Emission Factors (DEF) approach (calculated for the weighted mean only) and estimated with the crop model method (CM) for the 8 agronomic scenarios, minimum (Min) and maximum (Max) 144

Liste des abréviations

ACV	Analyse du Cycle de Vie / Life cycle Assessment (**LCA**)
AfricaRice	International agricultural research center for research on rice in Africa (Cotonou, Bénin)
AP	Acidification potential / Potentiel d'acidification
CEC	Capacité d'Echange Cationique (propriété du sol)
Cirad	Centre de coopération internationale en recherche agronomique pour le développement
CM	Crop Model (see method for Nr emission estimation) / Modèle biophysique
CML	Instititute of Environmental Sciences (Leiden, The Netherlands)
CS	Cropping System / Système de culture (**SdC**)
CV	Coefficient of Variation
DEF	Default Method (see method for Nr emission estimation)
ECETOC	European Center for Ecotoxicology and Technology of Chemicals
EF	Emission Factor / Facteur d'émissions
EP	Eutrophication Potential / Potentiel d'eutrophisation
ETP	Evapotranspiration
FAO	Food and Agriculture Organization of the United Nations
FC	Field Capacity / Capacité au champ (propriété du sol)
GHG	Greenhouse Gases / Gaz à effet de serre
GWP	Global Warming Potential/ Potentiel de réchauffement climatique
ICV	Inventaire du cycle de vie / Life cycle Inventory (**LCI**)
IFT	Indice de fréquence de Traitement / Treatment Frequency Index (**TFI**)
ILCD	International reference Life Cycle Data system (JRC, European Comission)
INRA	Institut National de Recherche Agronomique
INRAB	Institut National de Recherche Agronomique au Bénin
IPCC	International Panel for Climate Change
ISO	International Organization for Standardization
LCIA	Life Cycle Impact Assessment
NP	Engrais multiple à base d'azote et de phosphore
NPK	Engrais multiple à base d'azote, de phosphore et de potassium
N_r	Reactive nitrogen / Azote réactif
OECD	Organisation for Economic Co-operation and Development
SETAC	Society of Environmental Toxicology and Chemistry
STICS	Simulateur mulTIdisciplinaire pour les Cultures Standard (modèle biophysique)
UK	United Kingdom
WDP	Water Deprivation Potential / Potentiel de privation en eau
WFPS	Water-Filled Pore Space
WM	Weighted mean / Moyenne pondérée
WSI	Water Stress Index / Indice de stress hydrique

Introduction générale



Circulation urbaine et pollution à Cotonou, Bénin

Nourrir la planète : un enjeu global et des solutions locales

D'ici 2050, la population du globe devrait dépasser 9 milliards de personnes, soit environ deux milliards de plus qu'en 2010 (United Nations, 2012). Une augmentation de la production agricole est cruciale pour le maintien de la stabilité économique et sociale à l'échelle mondiale (Tilman et al., 2002). Dans les années 60-90, la révolution verte à grand renfort d'intrants a permis d'augmenter les rendements, mais l'agriculture d'aujourd'hui est dépendante de nombreuses ressources non renouvelables telles que les carburants fossiles, les minerais et l'eau et son empreinte écologique (dégradation des sols, pollution de l'eau, réduction de la biodiversité) a atteint un niveau sans précédent. Pourtant, les écosystèmes fournissent des services indispensables à la survie des populations les plus vulnérables et leur développement doit aller de pair avec la préservation des ressources naturelles et de l'environnement (Millennium Ecosystem Assessment, 2005). L'amélioration de la sécurité alimentaire doit donc passer par le développement durable de l'agriculture mondiale en général et dans les pays en développement en particulier. Malgré des progrès récents prometteurs (Pretty et al., 2003), la transition vers des systèmes de production plus durables dans les pays en développement reste un champ de recherche peu exploré, qui doit couvrir les produits de base, les produits d'origine animale et aussi les fruits et les légumes.

Dans les pays en développement, le maraichage est présent sous trois formes : (1) les cultures de rente destinées à l'exportation vers l'Europe comme le haricot du Kenya ou la tomate du Sénégal, (2) les cultures extensives en zone rurale pendant la saison des pluies et (3) les cultures intensives en zone urbaine et périurbaine tout au long de l'année. D'ici 2030, 60% de la population mondiale vivra en zone urbaine (United Nations, 2012). Dans ce contexte, le développement de l'agriculture urbaine représente un levier important pour le développement durable des villes dans les pays en développement en général, et en Afrique en particulier (FAO, 2012; World Bank, 2013). En effet, l'agriculture urbaine apporte des réponses pour réduire la pauvreté, améliorer la sécurité alimentaire, réduire la malnutrition et pallier les contraintes environnementales. Cependant, face à la demande croissante des urbains pour le foncier, les systèmes de production en zone urbaine se sont particulièrement intensifiés, mettant en évidence des risques environnementaux associés à ces modes de production (De Bon et al., 2010).

Une évaluation fiable de l'ensemble des impacts environnementaux associés aux systèmes de production maraichers en Afrique est aujourd'hui nécessaire.

Les méthodes d'évaluation de l'agriculture

Une grande diversité de méthodes a été développée pour l'évaluation de l'agriculture (van der Werf & Petit, 2002; Payraudeau & van der Werf, 2005; Bockstaller et al., 2009). L'objectif général de ces méthodes peut consister en l'évaluation environnementale stricte ou l'évaluation de la durabilité (performances économiques, environnementales et sociales). Ces méthodes permettent d'évaluer une variété d'impacts depuis l'échelle locale (odeur, destruction d'une zone protégée) jusqu'à l'échelle globale (changement climatique). Les critères ou indicateurs d'impacts peuvent être quantitatifs ou qualitatifs, simples ou multiples. Le type d'impact évalué peut être réel ou potentiel et être exprimé par une unité de référence unique ou multiple. Chaque méthode adresse plus particulièrement un niveau d'intégration spécifique de l'agriculture depuis la région jusqu'au système de culture en passant par la filière et l'exploitation agricole. Enfin, la pertinence d'une méthode d'évaluation repose sur la pertinence de ses indicateurs en lien avec les objectifs qu'elle s'est assignée. Il est ainsi préférable de choisir des indicateurs basés sur l'effet des pratiques (résultats) plutôt que sur la description des pratiques (moyens) permettant ainsi aux gestionnaires d'atteindre librement leurs objectifs en fonction de leur contexte de production (Payraudeau & van der Werf, 2005).

L'analyse des travaux sur les méthodes d'évaluation (Payraudeau & van der Werf, 2005) permet d'aboutir à un certain nombre de recommandations pour l'évaluation environnementale de l'agriculture que nous pouvons résumer ainsi :

- La prise en compte de l'ensemble des impacts environnementaux pertinents de type global ou local pour les systèmes de production étudiés, ainsi que de l'ensemble des étapes de production (site principal de production, production des intrants, commercialisation et consommation) permet de limiter le risque de transfert d'impact,
- La possibilité d'exprimer les impacts en fonction de plusieurs unités de référence permet la prise en compte de la multifonctionnalité des systèmes agricoles,
- L'évaluation quantifiée de l'effet des pratiques facilite la validation des résultats et permet une agrégation transparente qui préserve l'information concernant les points faibles des systèmes,
- Les concepts et indicateurs impliqués dans les méthodes doivent être validés scientifiquement et l'analyse de l'incertitude doit être intégrée dans le processus d'évaluation pour permettre une interprétation scientifiquement correcte des résultats.

En suivant cette grille de recommandations, la méthode Analyse du cycle de vie a été identifiée comme particulièrement pertinente pour l'évaluation environnementale des productions agricoles (van der Werf & Petit, 2002).

ACV et qualité des données d'inventaire

L'analyse du cycle de vie (ACV) est une méthode d'évaluation standardisée (ISO 14044, 2006) qui permet l'évaluation des impacts potentiels associés à un système de production. Ce système correspond au cycle de vie du produit, et inclut l'ensemble des processus depuis l'extraction des matières premières jusqu'à son utilisation ou sa destruction. On dit aussi « du berceau à la tombe ». La méthode ACV définit 3 aires de protection prioritaires : les ressources naturelles, la santé humaine et la santé des écosystèmes. A chaque zone de protection correspondent plusieurs catégories d'impact de type global comme le changement climatique ou de type régional comme l'eutrophisation.

La mise en œuvre de l'ACV comprend 4 étapes définies par la norme ISO 14044 (Figure 1). La première étape consiste à définir les objectifs et le champ de l'étude (ou périmètre). En pratique elle consiste à identifier le ou les systèmes étudiés par rapport à une fonction et à sélectionner une ou plusieurs unités de référence ou unités fonctionnelles grâce à laquelle l'ensemble des flux d'inventaire (étape 2) et les impacts (étape 3) seront exprimés. Cette étape permet également de délimiter les frontières du système et d'identifier les données et les méthodes à mettre en œuvre pour atteindre les

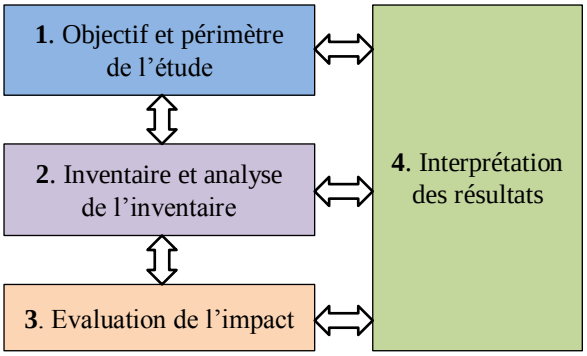


Figure 1 Relations entre les différentes étapes du processus d'analyse du cycle de vie (ISO, 2006**)**

objectifs de l'étude. L'étape suivante d'inventaire consiste à quantifier l'ensemble des flux entrants et des flux sortants du système. Ces flux seront ensuite convertis en impacts (étape 3) par un système d'équivalence entre les substances inventoriées et les substances de référence caractérisant chaque catégorie d'impact (le CO_2 pour le changement climatique). Enfin, la phase d'interprétation consiste à évaluer la valeur et la robustesse de tous les résultats, choix et hypothèses en relation avec les objectifs initiaux. La méthodologie ACV prévoit une boucle itérative pour permettre le raffinement des étapes 1, 2 ou 3 si les objectifs ne sont pas atteints.

Dans l'analyse du cycle de vie comme dans toute approche de modélisation, la qualité des données est l'une des principales limites de l'application de la méthodologie. La variabilité des mesures entre les systèmes de production, les divergences entre la bibliographie et les données de terrain et la sensibilité des résultats aux choix méthodologiques constituent les principaux obstacles à la fiabilité et à la comparabilité des résultats d'ACV (Coulon et al., 1997). Ces préoccupations sont au cœur des enjeux scientifiques liés à l'application de l'ACV aux produits agricoles.

Empruntées au champ disciplinaire de l'agronomie, deux thématiques ont donné lieu à des développements méthodologiques récents en ACV : la prise en compte de la diversité des systèmes de production et de la variabilité des émissions.

Diversité des systèmes de production

La diversité des systèmes de production agricoles (ou fermes) résulte de décisions techniques et stratégiques prises par les agriculteurs pour combiner leurs facteurs de production (moyens disponibles) et leurs modes de production (objectifs de production). Chaque situation étant unique, cette diversité est quasiment infinie. Or, la diversité de ses fermes est considérée comme l'une des richesses d'une agriculture, comme un gage de sa capacité d'adaptation, et le maintien de cette diversité est de plus en plus souvent évoqué comme un objectif en soi (Perrot & Landais, 1993).

La typologie a largement été utilisée pour décrire et structurer des populations de fermes pour répondre à des besoins d'analyse et d'évaluation de l'agriculture. Historiquement, les méthodes de typologie ont été développées et mises en œuvre (notamment en France) dans un objectif de développement rural pour évaluer le fonctionnement des fermes, trouver des solutions aux problèmes rencontrés et élaborer des recommandations adaptées (Capillon, 1993; Landais, 1998). Plus largement, la typologie permet l'analyse d'une situation donnée définie d'un point de vue géographique (Köbrich

et al., 2003; Kristensen & Kristensen, 2004) et/ou technique (Tittonell et al., 2005; Maton et al., 2005; Poussin et al., 2008). Les méthodes de typologie ont également été mises en œuvre pour concevoir des systèmes de production innovants (Blazy et al., 2009) et analyser les déterminants de leur adoption (Daskalopoulou & Petrou, 2002; Bidogeza et al., 2009). Enfin, les méthodes de typologie peuvent être à la base d'approche ex-ante, pour analyser l'impact d'un changement de l'environnement biophysique ou politique (Smit & Skinner, 2002; Andersen et al., 2007).

De manière générale, les méthodes de typologie consistent à élaborer des types pour analyser une réalité complexe et ordonner des objets relativement similaires. Cette structuration plus ou moins artificielle est basée sur des critères de distinction, qui découlent de l'identification des principales causes explicatives de la diversité (Andersen et al., 2007). En pratique, l'objectif est de maximiser l'homogénéité à l'intérieur d'un type particulier tout en maximisant l'hétérogénéité entre les types (Köbrich et al., 2003). Une large gamme de méthodes a été développée : les approches les plus qualitatives et supervisées (*a priori*) facilitent une analyse cognitive et fonctionnelle des fermes d'autant plus qu'elles intègrent des notions de stratégie et de trajectoire temporelle. Ces approches nécessitant des enquêtes approfondies et une analyse fastidieuse, sont souvent limitées à de petites régions dans lesquelles l'agriculture est organisée. Les approches les plus quantitatives et automatisées faisant appel aux méthodes statistiques (analyse en composante principale et classification hiérarchique) permettent l'analyse de populations plus larges, pourvu qu'un grand nombre de données soit disponible. Cependant ces approches sont très dépendantes de la représentativité de l'échantillon et des variables sélectionnées. En pratique, les méthodes sont très souvent semi-supervisées de manière plus ou moins générique, et incluent à la fois des variables quantitative et des indicateurs qualitatifs.

L'approche par scénarisation développée par Basset-Mens et al. (2007) et l'approche de typologie de fermes développée par Dalgaard et al. (2006) sont deux exemples de développement récents pour améliorer la prise en compte de la diversité des systèmes de production dans le cadre de l'analyse du cycle de vie. Les deux approches se basent sur des données mixtes issues de bases de données et d'expertise. L'approche de Basset-Mens et al. (2007) distingue les systèmes sur la base du mode de production (quels objectifs de production ?) alors que l'approche de Dalgaard et al. (2006) distingue les systèmes sur la base des facteurs de production (quels moyens mis en œuvre ?).

Variabilité des émissions au champ

Les études relatives aux émissions gazeuses (NH_3, N_2O, NO_x, N_2) s'accordent sur la grande variabilité des émissions en fonction des pratiques et du contexte pédoclimatique, surtout à très petites échelles (parcelle). Par exemple, Bussink et Oenema (1998) ont montré que le coefficient de variation associé aux émissions de NH_3 associées à l'application de fumier peut atteindre 100%. Une analyse bibliographique portant sur des données mesurées et la modélisation des émissions de N_2O annuelles à

l'échelle globale (Stehfest & Bouwman, 2006) a conclu qu'aucune amélioration n'avait été réalisée récemment concernant l'évaluation de N_2O à l'échelle globale. Un grand nombre de facteurs influant sur les émissions d'azote réactif (NH_3, N_2O, NO_x, NO_3) a été mis en évidence dans la littérature spécifique à ces flux (Del Grosso et al., 2000; Sommer & Hutchings, 2001; Milroy et al., 2008). Ces facteurs d'influence résultent de la combinaison entre les caractéristiques de sol, de climat et de pratiques agricoles des systèmes de production et des processus biophysiques, ayant un effet direct sur les émissions.

Contrairement aux systèmes de production clos comme les usines pour lesquels l'ACV a initialement été développée, il est difficile de mesurer les émissions à la sortie des processus de production en agriculture. D'une part parce que ces émissions sont majoritairement de nature diffuse dans le temps et l'espace et d'autre part parce qu'elles résultent de processus biologiques complexes. Dans le champ de l'ACV, le recours aux modèles stochastiques (Payraudeau et al., 2007) ou biophysiques (Gabrielle & Gagnaire, 2008; Langevin, 2010; Bessou et al., 2013; Dufossé et al., 2013) sont deux exemples de développement récents pour améliorer la prise en compte de la variabilité des émissions d'azote associées à l'usage d'engrais dans les études ACV.

Evaluation environnementale de la tomate produite en agriculture urbaine au Bénin

L'horticulture urbaine est une des activités pratiquées par les agriculteurs urbains. Au même titre que l'agriculture urbaine elle se caractérise par une recherche de valorisation forte des facteurs de production dans un contexte de forte pression foncière, de besoin en main d'œuvre, d'accès limité aux intrants (De Bon et al., 2010). Les cultures horticoles en général et la tomate en particulier sont très sensibles aux aléas climatiques et à la pression phytosanitaire. Elles nécessitent des investissements importants pour les infrastructures d'irrigation et les intrants. A cause du faible niveau d'éducation des producteurs et de leur accès limité au soutien technique, les producteurs ont tendance à apporter des intrants de manière excessive par rapport à leurs niveaux de rendement. Les perspectives de développement de l'horticulture urbaine soulèvent l'existence de risques environnementaux associés à un usage mal maitrisé des intrants (Orsini et al., 2013). Plusieurs études ont notamment relevé des pratiques de fertilisation intensive entrainant des surplus azotés importants (Diogo et al., 2010; Predotova et al., 2010; Predotova et al., 2011). D'autres études ont décrit des pratiques de protection des cultures intensives, avec un usage excessif de pesticides (Cissé et al., 2003; Ahouangninou et al., 2012).

Le cas de la tomate en jardin urbain au Bénin est particulièrement illustratif de cette problématique. Le Bénin présente une densité de population deux fois plus élevée par rapport à la moyenne africaine (87 vs. 37 hab.km-²). La population croit deux fois plus vite en zone urbaine qu'en zone rurale (4 vs.2%) et 6% de la population est en situation de malnutrition. La tomate est la première production horticole du pays (FAO, 2013). Malgré tout, les villes sont obligées d'importer des tomates en saison sèche et

cela malgré les 400 ha additionnels alloués récemment à l'agriculture urbaine dans les communes de Cotonou et Sèmè-Kpodji (FAO, 2012). La principale raison évoquée par la littérature grise pour expliquer ce défaut de production est le manque de maitrise technique pour faire face à des conditions climatiques chaudes et humides et à des attaques de ravageurs en tout genre. Ainsi, le développement de l'horticulture urbaine en général et de la production de tomate en jardin urbain au Bénin en particulier soulève de nouvelles questions agronomiques et environnementales. Des études et outils d'évaluation environnementale fiables de ces systèmes sont nécessaires pour y répondre et accompagner un développement durable de ces systèmes.

L'application de méthodologies de type ACV, très exigeantes en données, aux systèmes maraîchers urbains, représente cependant un défi méthodologique important au regard des très faibles disponibilités en données et en connaissances sur ces systèmes.

Contexte et objectifs de la thèse

Dans un contexte d'urbanisation croissante, les décideurs politiques, les scientifiques et les acteurs locaux sont interpelés pour protéger et promouvoir l'agriculture urbaine (et péri-urbaine), parce qu'elle contribue à la fois à satisfaire les besoins d'alimentation, d'emploi, et de traitement des déchets (De Bon et al., 2010; Orsini et al., 2013). Pour concevoir des systèmes plus productifs, durables, économes en intrants et moins nocifs pour l'environnement, il est nécessaire d'identifier des outils sur lesquels s'appuyer pour faire des choix techniques pertinents. Cette thèse s'intègre dans un dispositif mis en place par le Cirad depuis 2009 sur le thème de l'analyse de cycle de vie des produits tropicaux, qui comprend une équipe de chercheurs spécialistes de ce nouveau champ de recherche, des outils de calcul et des bases de données mutualisées.

La question scientifique posée dans cette thèse peut être formulée ainsi : comment prendre en compte la variabilité pédoclimatique et technique dans la réalisation des inventaires environnementaux et ainsi affiner les résultats d'analyse du cycle de vie des produits maraîchers en contexte tropical? Sur la base de cette question, plusieurs objectifs sont formulés :

- **Objectif 1** : réaliser l'état de l'art de l'ACV sur produits maraîchers afin de concevoir un protocole d'étude adapté
- **Objectif 2** : explorer les approches possibles pour mieux caractériser la diversité des systèmes de culture au regard de leurs impacts environnementaux, et créer des inventaires représentatifs,
- **Objectif 3** : identifier les déterminants de la variabilité des émissions au champ et explorer les approches possibles, telles que la modélisation, pour raffiner les méthodes d'estimation des émissions azotées,
- **Objectif 4** : évaluer l'impact environnemental de la tomate produite en jardins urbains au Bénin et
- **Objectif 5** : produire des recommandations pour améliorer la qualité des résultats d'ACV pour les produits maraichers tropicaux.

Plan de la thèse et démarche scientifique

Ces objectifs nous ont amené à construire une démarche de travail cohérente, reprise dans la structuration de la thèse (Figure 2), qui s'articule autour des 4 étapes de l'analyse du cycle de vie présentées en figure 1. En préalable à notre propre contribution, une revue de la littérature doit nous permettre d'identifier les grands enjeux méthodologiques de l'application de l'ACV aux produits maraîchers (chapitre 1). L'étape de l'inventaire fera l'objet de développements méthodologiques spécifiques pour répondre aux objectifs 2 (chapitre 2) et 3 (chapitre 3) explorant respectivement les apports de la typologie puis de la modélisation biophysique. L'étape d'évaluation des impacts (objectif 4) sera construite autour de scénarios et présentée au chapitre 4. L'étape d'interprétation des résultats sera approfondie d'abord sur la base des résultats d'ACV présentés dans les études publiées de notre revue (chapitre 1) puis sur la base des résultats d'ACV pour le cas d'étude tomate Bénin (chapitre 5). Enfin, des recommandations seront formulées (objectif 5) suite à l'analyse de l'ensemble de ces tentatives exploratoires dans la discussion générale du document de thèse.

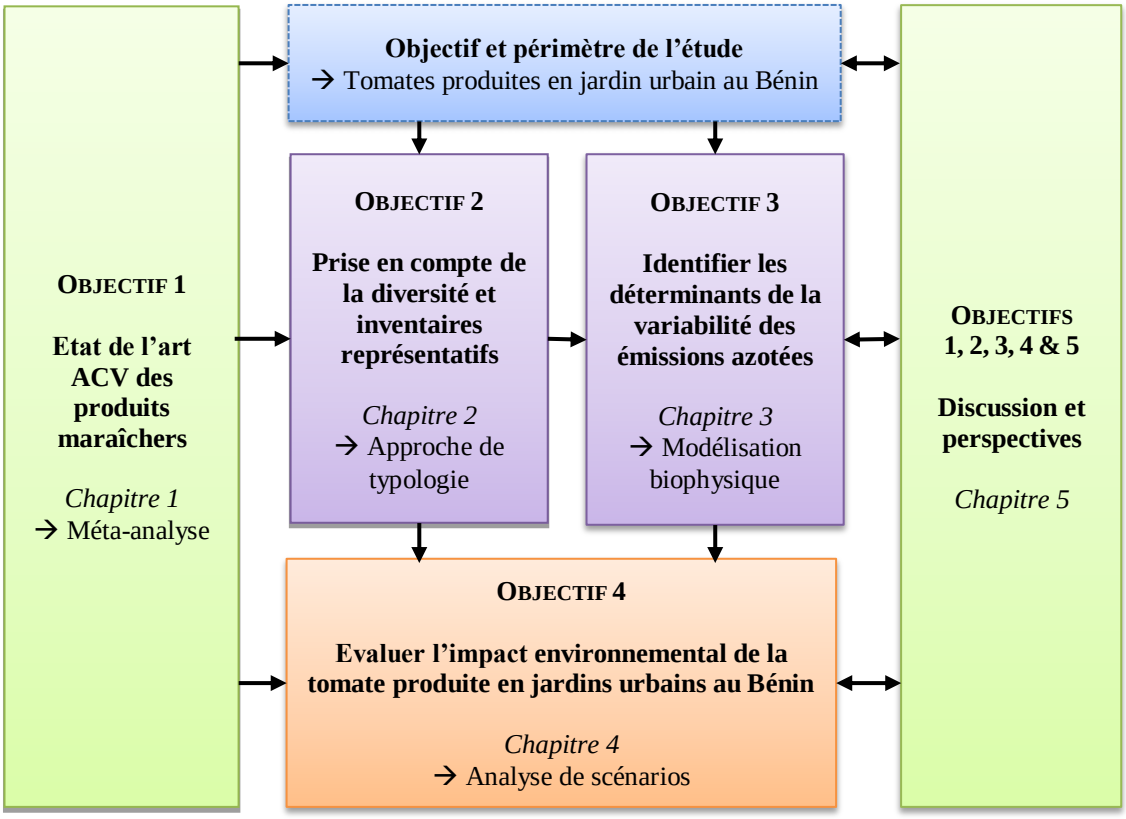


Figure 2 Plan de la thèse et démarche scientifique

Le chapitre 1 présente une revue de la littérature sur l'analyse du cycle de vie appliquée aux produits maraîchers. Grâce à une approche de type méta-analyse, nous avons analysé la qualité des données et des méthodes d'inventaire et leurs effets sur la variation des résultats d'ACV. Nous avons ainsi

formulé des recommandations pour la réalisation d'ACV plus fiables et plus comparables pour l'évaluation environnementale des produits maraichers.

Fort de ce constat général, nous avons identifié un cas d'étude pertinent par rapport aux enjeux identifiés pour notre travail et sur lequel mettre en œuvre nos recommandations : la tomate produite en jardin urbain au Bénin. La méthodologie ACV est particulièrement pertinente pour aborder les problématiques environnementales associées à l'intensification des systèmes de production en zone urbaine et péri-urbaine et plus particulièrement à leur usage excessif de pesticides et d'engrais. De plus, le manque de données et de connaissances sur ces systèmes accentue les défis scientifiques associés à la qualité des inventaires.

Au chapitre 2 nous présentons une procédure de création d'inventaires représentatifs basée sur la typologie. Cette approche permet de caractériser la diversité des systèmes de production d'un point de vue agronomique et de quantifier les flux d'inventaires pour la réalisation d'ACV.

Le chapitre 3 présente notre démarche d'exploration de la variabilité des émissions d'azote au champ reposant sur l'usage d'un modèle biophysique combiné à une approche de bilan. A partir des données de terrain présentées dans le chapitre précédent, nous mettons en évidence les principaux facteurs influents des émissions d'azote dans notre cas d'étude et discutons les performances et les limites de ce type d'approche pour raffiner l'estimation des émissions.

Au chapitre 4, nous intégrons les principaux résultats des deux chapitres précédents pour réaliser la première ACV détaillée de la tomate en maraichage urbain au Bénin. Sont présentés dans ce chapitre une gamme de profils environnementaux représentatifs de la diversité des systèmes de culture étudiés. Une analyse de sensibilité des résultats d'ACV à la variabilité des émissions d'azote est également proposée sur la base de 8 scénarios agronomiques.

Enfin, **le chapitre 5** est une discussion générale de l'ensemble des résultats de la thèse. Il s'articule autour de la discussion des résultats et méthodes testées pour atteindre les quatre premiers objectifs de la thèse et permet ultimement d'atteindre l'objectif 5 : en formulant des recommandations pour améliorer l'évaluation environnementale des systèmes de production de tomate en maraîchage urbain et plus généralement des produits horticoles tropicaux.

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Chapitre 1. Analyse du cycle de vie des systèmes de culture maraichers: une revue centrée sur la diversité des systèmes de culture et l'estimation des émissions au champ



Parcelle de choux dans un jardin urbain à Cotonou, Bénin

Avant-propos

Avant de se lancer dans la réalisation de l'ACV pour notre cas d'étude nous avons cherché à identifier des précédents dans la littérature. Or au début de ce travail de thèse, la littérature ACV appliquée à des produits maraîchers tropicaux se limitait à quelques cas de produits importés vers l'Europe et contenait très peu de considérations méthodologiques. Parallèlement, la littérature ACV appliquée aux produits maraîchers abordait une large gamme de systèmes de production mais les pratiques d'application de l'ACV à ces produits étaient très variables et ne bénéficiaient pas de règles de bonnes pratiques couvrant l'ensemble des particularités associées à ces systèmes de production.

A partir de ce constat, il était nécessaire de réaliser l'état de l'art des pratiques actuelles en ACV des produits maraîchers pour identifier et évaluer les pratiques actuelles, les verrous méthodologiques, les sources de variabilité et d'incertitude sur lesquels se baser pour construire notre propre protocole dans notre cas d'étude mais aussi pour formuler des recommandations pour améliorer la qualité et la comparabilité des études ACV des produits maraîchers. C'est l'objet du papier de revue de la littérature qui suit et qui a été publié dans International Journal of LCA.

Résumé

Les récentes études d'Analyse de Cycle de Vie (ACV) des fruits et légumes mettent en évidence que la phase de production agricole concentre une part prépondérante des impacts environnementaux de ces produits. D'autre part, les systèmes de culture maraichers sont très divers selon les zones géographiques et les systèmes de production. Ainsi, cette revue bibliographique s'attache à analyser les méthodes et les données d'inventaires utilisées dans les différentes études ACV et à comparer les impacts environnementaux potentiels de différentes productions maraîchères au stade de la ferme.

Dix études publiées dans des revues à comité de lecture ou dans des rapports répondant aux normes ISO ont été sélectionnées. Nous avons d'abord analysé qualitativement les objectifs et le cadre des différentes études. Ensuite, les résultats d'inventaires et les résultats d'impact pour trois catégories (changement climatique, eutrophisation, acidification) ont été analysés au moyen d'une typologie des systèmes de production, mais aussi par groupe de produit et par étude. Enfin, une attention particulière a été portée sur les méthodes d'estimation des émissions d'azote réactif.

Parmi les différents systèmes de cultures identifiés, les systèmes sous serres chauffées présentent le potentiel de changement climatique le plus élevé en raison de leur consommation d'énergie pour le chauffage. Entre les différents groupes de produits, les haricots présentent les potentiels d'acidification et d'eutrophisation les plus forts en raison de fortes émissions dues aux importantes quantités d'engrais utilisées par tonne de produit. En analysant les résultats par étude, il ressort que les principales sources de variation d'impact sont les rendements, les pratiques de fertilisation et les choix méthodologiques relatifs aux règles d'allocation. Globalement, la comparabilité des études ACV pour les productions maraîchères est restreinte par (1) le manque de transparence sur les objectifs et périmètres des analyses, (2) le manque de données représentatives pour décrire les systèmes de production et (3) l'hétérogénéité et le manque de spécificité des méthodes pour l'estimation des émissions aux champs dans les conditions de production maraîchères. En particulier, il apparait que les méthodes d'estimation des émissions d'azote sont souvent appliquées en dehors de leur domaine de validité.

Cette première tentative de comparaison des impacts potentiels des productions maraichères a mis en évidence plusieurs manques en termes de données et de méthodes dans la perspective de produire des résultats représentatifs pour la phase de production agricole des produits maraîchers. Ainsi, une série de recommandations a été formulée afin de mieux prendre en compte les spécificités des productions maraichères, et ainsi d'améliorer la qualité des études ACV sur ces produits.

Mots clés : Stade ferme, maraîchage, typologie, méthodes d'estimation des émissions, azote réactif.

Life cycle assessment of vegetable products: a review focusing on cropping systems diversity and the estimation of field emissions

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Abstract

Purpose Recent life cycle assessment studies for vegetable products have identified the agricultural stage as one of the most important contributors to the environmental impacts for these products while vegetable production systems are characterized by specific but also widely diverse production conditions. In this context, a review aiming at comparing the potential impacts of vegetable products and analysing the relevance of the methods and data used for the inventory of the farm stage appeared necessary.

Method Ten papers published in peer-reviewed scientific journals or ISO-compliant reports were selected. First, a presentation of the selected papers was done to compare the goal and scope and the life cycle inventory data to the related sections in the ILCD Handbook. Second, a quantitative review of input flows and Life Cycle Impact Assessment (LCIA) results (global warming, eutrophication and acidification) was based on a cropping system typology and on a classification per product group. Third, an in-depth analysis of the methods used to estimate field emissions of reactive nitrogen was proposed.

Results The heated greenhouse system types showed the greatest global warming potential. The giant bean group showed the greatest acidification and eutrophication potentials per kilogram of product while, the tomato group showed the greatest acidification and eutrophication potentials per unit of area. Main sources of variations for impacts across systems were yields and inputs variations and, system expansion rules. Overall, the ability to compare the environmental impact for these diverse vegetable products from cradle-to-harvest was hampered by (1) weaknesses regarding transparency of goal and scope, (2) a lack of representativeness and completeness of data used for the field stage and (3) heterogeneous and inadequate methods for estimating field emissions. In particular, methods to estimate reactive nitrogen emissions were applied beyond their validity domain.

Conclusions and recommendations This first attempt at comparing the potential impacts of vegetable products pinpointed several gaps in terms of data and methods to reach representative LCIA results for the field production stage. To better account for the specificities of vegetable cropping systems and improve the overall quality of their LCA studies, our key recommendations were: (i) to include systematically phosphorus, water and pesticide fluxes and characterise associated impacts: eutrophication, toxicity, water deprivation, (ii) to better address space and time representativeness for field stage inventory data through better sampling procedures and reporting transparency and, (iii) to use best available methods and when possible more mechanistic tools for estimating Nr emissions.

Keywords: *Vegetable production; typology of cropping systems; emissions; Life Cycle Assessment; reactive nitrogen; greenhouse; open-field systems*

1.1. Introduction

The year-round demand for a diversity of vegetables and fruits from Western markets has led to the development of complex supply chains (farming, grading, packing, transport, storage, refrigeration, retailing…) often spread across distant countries. Particularly for off-season supply, this demand is also responsible for the development of artificial cropping systems under temperate climate (soilless and heated greenhouses) and of a range of cropping systems (from soilless and sheltered to open-field productions) in Mediterranean and tropical countries. The growing awareness regarding the environmental issues due to global food supply chains has now reached the fruits and vegetables sector and the need for evaluating the different product origins and associated technologies has clearly been expressed.

The Life Cycle Assessment (LCA) methodology was commonly used to assess the environmental performance of agricultural products over the last fifteen years, revealing several scientific challenges due in particular to the use of land by farming systems, their interactions with their environment, their large variation in terms of soil, climate and practices and the importance of local and regional impacts (Brentrup et al., 2004). Although the relevance of LCA to assess agricultural products was validated for a large number of agricultural products (Brentrup et al., 2001; Mila i Canals et al., 2006), its application to horticultural (i.e. fruits and vegetables) production systems was more recent and came with renewed and specific scientific challenges.

Since horticultural products and production systems are characterized by specific conditions of production and of interaction with their environment, but also by a large diversity of technologies and management practices, a review of the state of the art of LCA studies for horticultural products, as a recent object of study, appears necessary and timely. A recent review focusing on perennial crops (Bessou et al., 2013) already provided important insights regarding specificities of LCA applied to orchards (including fruit production). No relevant analysis has been made on LCA applied to annual and even shorter crop length products, e.g. vegetables. In available LCA studies of vegetable products, contribution analyses from cradle-to-grave and from-cradle-to-retail studies revealed a significant contribution of post-harvest stages (Andersson et al., 1998; Sim et al., 2007). However, authors widely agreed upon the field production stage being one of the greatest contributors especially to global warming, acidification, eutrophication and toxicity potentials (Andersson et al., 1998; Antón et al., 2005a; Mila i Canals et al., 2008; Hospido et al., 2009; Cellura et al., 2012b). The importance and complexity of the field production stage in the LCA of vegetable products requires putting a particular emphasis on the analysis of the methods and data used for this stage.

Vegetable cropping systems show a huge diversity of farming and environmental conditions of production but also important specificities that largely impact the inventory, the associated on-field

estimation methods and the resulting LCA outcomes. Although this was already pinpointed in the literature, the methods and data used for the description of vegetable products and the calculation of their environmental inventory have never been reviewed nor have been their implications for LCA results. We focus in this paper on the cradle-to-harvest stages, including the production of agricultural inputs and the associated emissions. The objectives of this review were three-fold:

1. To assess the state of the art of published LCA studies on vegetables including an original typology-based method, with a focus on data and methods used for the field production stage
2. To carry out a critical analysis of the methods used for the estimation of reactive nitrogen (Nr) emissions
3. To make recommendations in terms of practice and scientific challenges for applying LCA to vegetable cropping systems

1.2. Materials and methods

1.2.1. Selection of LCA studies and review process

A thorough literature search resulted in the identification of seventeen papers published in peer-reviewed scientific journals or in ISO-compliant reports, which presented full LCA studies for vegetable products. Studies which did not involve multi-criteria assessment (mainly those studies focused on energy and global warming potential) were excluded from our review, as well as reports which were not accessible on the Internet. A subsample of 10 studies featuring consistent and well-described data for the modeling of cropping systems was found suitable for a qualitative and quantitative analysis.

Our qualitative review of goals and scopes (section 1.3.1) covered the main aspects described in the recent ILCD handbook related sections (European Commission, 2010). Our quantitative review of LCI (sections 1.3.2 and 1.3.3) and LCIA results (section 1.3.4) were based on an original typology described in the next section (section 1.2.2). Results were also presented according to their product group and per reference. Across selected studies, only Global Warming Potential, Eutrophication potential and Acidification potential were homogeneously calculated following the CML methodology (Guinée et al., 2001) and could be selected for the quantitative analysis. The critical analysis (sections 1.4.1 to 1.4.3) followed the main recommendations from the ILCD handbook. An in-depth analysis was also made on methods used for the estimation of nitrogen reactive emissions (section 1.4.4).

1.2.2. Typology of cropping systems and aggregation of results

The cropping system generally results from the combination of a product's choice (targeted market), certain objectives of production (yield and quality) and a location. For vegetable products, this leads to diverse degree of artificialization in terms of protection status, heating and growing media to overcome local natural constraints and reach the production objectives. To explore the cropping

systems diversity from our panel of papers we developed a typology of cropping systems, based on three main criteria: protection and heating status (heated or cold greenhouse or open-field), growing media (substrate or soil) and climate (temperate, Mediterranean or tropical context). This typology was relevant to LCA as energy consumption is closely related to the greenhouse type and management (heated or not), while the growing media and the climate characteristics affect inputs rates, yields and field emissions. This typology resulted in 7 types named according to their characteristics. Greenhouse systems are either heated (GH-heat) or cold (GH-cold). Both systems can be led soilless on substrate (eg. GH-heat-sub) or directly on soil (eg. GH-heat-soil). Open-field cropping systems were classified according to their respective climate: temperate (OF-Temp), Mediterranean (OF-Med) or tropical (OF-Trop). Overall, 72 cropping systems were analysed and grouped into 7 cropping system types.

Results were also aggregated by product group. Indeed, crop species have specific management requirements and potential yields, which affect their environmental performances. Across the 10 studies, 7 product groups were established namely, giant bean (Abeliotis et al., 2013), green bean (Mila i Canals et al., 2008; Romero-Gámez et al., 2012), broccoli (Mila i Canals et al., 2008), cauliflower (Martínez-Blanco et al., 2010), leek (de Backer et al., 2009), salad (Mila i Canals et al., 2008) and tomato (Anton et al., 2004; Martínez-Blanco et al., 2009; Boulard et al., 2011; Martínez-Blanco et al., 2011; Torrellas et al., 2012). Finally, we presented the results per reference to analyze possible variations due to particular methodological choices by authors.

1.3. Presentation of LCA studies selected

1.3.1. Goal and scope

Intended applications

We can distinguish 3 objectives of the LCA studies reviewed:

- Comparing different farming practices (Romero-Gámez et al. 2012; Torrellas et al. 2012; Martínez-Blanco et al. 2011; Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2009; de Backer et al. 2009, Antón et al., 2005a)
- Comparing different product‘s origins such locally-grown products versus imported equivalent (Mila i Canals et al. 2008)
- Producing a reference for the environmental performance of a product in a given context (Abeliotis et al. 2013; Boulard et al. 2011).

Function and functional unit

Most papers studied the function of production expressed in mass of fresh or dry product (Abeliotis et al. 2013; Torrellas et al. 2012; Boulard et al. 2011; de Backer et al. 2009; Antón et al. 2005a). The production function could be associated to quality aspects either from a commercial standard (Romero-Gámez et al. 2012; Martínez-Blanco et al. 2011; Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2009) or from a nutritional value perspective (Martínez-Blanco et al. 2010). The functions studied across those papers were often multiple including land occupation functions (Abeliotis et al. 2013; de Backer et al. 2009) and could refer to functions beyond the production, such as the consumption expressed in kilograms of vegetable consumed (Mila i Canals et al. 2008).

Comparison of systems

Across the 10 reviewed papers, contrasted vegetable systems were studied. All studies had in common to present original inventories related to field production stages of vegetable products. Indeed, each cropping system was unique in terms of management and yield. These management and yield variations resulted from different modalities in terms of cropping system characteristics:

- The protection status: greenhouse versus open-field (Romero-Gámez et al. 2012; Martínez-Blanco et al. 2011; Mila i Canals et al. 2008), greenhouse type (Torrellas et al. 2012; Boulard et al. 2011) and greenhouse management (Torrellas et al. 2012; Romero-Gámez et al. 2012; Antón et al. 2005a),
- The growing media: substrate versus soil (Boulard et al. 2011; Antón et al. 2005a),
- The location: temperate, Mediterranean or tropical contexts (Mila i Canals et al. 2008),
- The fertilization management, involving different types of fertilizer (mineral fertilizer versus compost) and application rates (Martínez-Blanco et al. 2011; Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2009).
- The production orientation: conventional, integrated or organic (de Backer et al. 2009; Abeliotis et al. 2013),
- The product cultivar (Abeliotis et al. 2013).

Finally, management and yield variations also resulted from between-farms and within-year variability (Mila i Canals et al. 2008).

System boundaries and cut-off criteria

Reflecting those various functions but also the diversity of vegetable cropping systems studied, the system limits were diverse. As a rule, the system boundaries for cradle-to-farm-gate LCA included agricultural operations (including greenhouse management) and on-field emissions due to fertilizers and pesticides within the foreground system ("foreground" refers here to the farm stage) and the

production, transport and use of buildings, machineries and agricultural inputs within the background system ("background" refers to processes before the farm stage). Several additional stages were recorded across studies. Eight studies explicitly included the waste treatment phase (Torrellas et al. 2012; Romero-Gámez et al. 2012; Boulard et al. 2011; Martínez-Blanco et al. 2011; Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2009; Antón et al. 2005a). Six studies explicitly included the nursery phase (Torrellas et al. 2012; Boulard et al. 2011; Martínez-Blanco et al. 2011; de Backer et al. 2009; Antón et al. 2005a). Boulard et al. (2011) included the packaging phase occurring on the farm and Mila i Canals et al. (2008) included fuel consumption related to the transport of workers. Some studies included the biogenic fixation of atmospheric carbon dioxide in plants (Mila i Canals et al. 2008) or in soil after compost application (Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2011). We also noticed some punctual omissions, such as transport and packaging of agricultural inputs (Romero-Gámez et al. 2012), building and machinery production and transport (Abeliotis et al. 2013; de Backer et al. 2009), pesticides production and associated field-emissions (Torrellas et al. 2012; Martínez-Blanco et al. 2011; Martínez-Blanco et al. 2010) or quite often simply pesticides field-emissions (Abeliotis et al. 2013; Romero-Gámez et al. 2012; Martínez-Blanco et al. 2009).

Several procedures were described regarding multifunctional processes and flows crossing the cropping system limits. Martínez-Blanco et al. (2009) and following papers (Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2011) assigned avoided burdens from dumping organic wastes to their system using compost. Torrellas et al. (2012) presented two scenarios regarding the expansion of their heated greenhouse tomato system to co-generation of heat and electricity: (1) avoided burden from producing the electricity used for the system using co-generation, (2) allocation of gas consumption for greenhouse heating based on energy criteria. In Martínez-Blanco et al. (2011) and in Boulard et al. (2011), the allocations of compost and amendments were based on plant uptake. The allocations of infrastructure over annual rotations were based either on economic criteria (Boulard et al. 2011) or on occupation time criteria (Mila i Canals et al. 2008).

1.3.2. Data and methods for LCI

Type, origin and quality of data

Across selected studies, type and origin of foreground data was of a wide diversity. Agricultural practices data were always presented as site-specific, i.e. representative for one vegetable product at the regional or the country scale. However, those data sets varied in time and space scales from national production data recorded for one or a few years, to individual field data for a given crop cycle defined according to its location, its duration and its position during the year. Data sources were also very variable: Boulard et al. (2011) and de Backer et al. (2009) used data from large-scale surveys carried out by agricultural development and extension services, while Abeliotis et al. (2013) and Mila i Canals et al. (2008) used data based on a restricted number of farms surveyed specifically for their

study. Martínez-Blanco et al. (2009; 2010; 2011) and Romero-Gámez et al. (2012) used data from a 1 or 2-year trial, whose management followed regional recommendations. Torrellas et al. (2012) used a mix of data collected by local partners, producers and sales cooperatives, as well as region-specific literature and data from experimental stations. Antón et al. (2005a) used a mix of data from a farmers' association for the soil cultivation scenario and from trials for the soilless scenarios. Background data was quite homogeneous among the panel of 10 studies. Data regarding buildings, machinery and inputs production were mainly based on literature or European and international databases. One exception was Martínez-Blanco et al. (2009) who provided site-specific data regarding compost production, based on data collected from a local compost plant.

Methods for field emissions

Table 1.1 shows the guidelines or references (column 1 and 2) used by authors (last column) to estimate field emissions for various forms of reactive nitrogen (N_r). Most studies referred to recommendations from two general guidelines (Audsley et al., 1997; Nemecek & Kägi, 2007) and one original paper (Brentrup et al., 2000), although transparency issues arose. Abeliotis et al. (2013) did not explicitly state whether they included the emissions from mineral fertilizers or not. Torrellas et al. (2012), Martínez-Blanco et al. (2011) and Martínez-Blanco et al. (2010) referred to several guidelines without specifying which one was used for each type of N_r emission.

Some adaptations were explicitly described such as the extension to organic fertilizers of the recommendation from Audsley et al. (1997) regarding nitrous oxide (N_2O) emissions (Mila i Canals et al., 2007; Mila i Canals et al., 2008). A few methods more specific to studied systems were also mentioned: Boulard et al. (2011) used publications specific to greenhouse production systems to estimate nitrate (NO_3) emissions distinctly from open and closed-loop systems, while Torrellas et al. (2012) deliberately omitted NO_3 emissions from systems involving a recirculation of drainage water.

The calculation of phosphorus (P) emissions was only detailed in 4 studies. Abeliotis et al. (2013) referred to recommendations from Nemecek and Kägi (2007) and de Backer et al. (2009) referred to Audsley et al. (1997). Boulard et al. (2011) and Mila i Canals et al. (2007; 2008) used data from specific publications, being respectively Sedilot et al.(2002) and Cowell and Clift (1998). Finally, Antón et al. (2005a) deliberately omitted PO_4 emissions on the basis that phosphorus was immobilized in the topsoil due to its high pH.

Regarding pesticides emissions, only 3 studies explicitly mentioned their inclusion in the inventory. Boulard et al. (2011) and Antón et al. (2005a) referred to the model from Hauschild (2000) adapted by Antón et al.(2004). De Backer et al. (2009) referred to the model from Birkved and Hauschild (2006).

Finally, only Mila i Canals et al. (2008) explicitly included CH_4 emissions from ammonium fertilizer application in the inventory.

Table 1.1 LCI methods to estimate reactive nitrogen field emissions: guidelines and references, calculation method using emission factors and parameters, validity domain and uses in selected studies

Guideline	**References**	**Value, emission factors (EF) - parameters**	**Validity domain of the reference** *Cropping system*	*Location*	**Uses in papers** [a]
NH_3 emissions from mineral fertilizers					
Audsley et al., 1997 Nemecek and Kägi, 2007	Asman, 1992	**EF** % of $N_{fertilizer}$ - Fertilizer type	Laboratory experiment	Europe	2, 4, 7, 9, 10_b (1, 3, 5, 6)
Brentrup *et al.*, 2000	ECETOC, 1994	**EF** % of $N_{fertilizer}$ - Fertilizer type - Soil properties (pH, $CaCO_3$)	Mineral fertilizer	European Countries	8, (3, 5, 6)
NH_3 emissions from organic fertilizers					
Audsley *et al.*, 1997	Menzi et al., 1998_b	**EF= 50%** of $N\text{-}NH_{4\ applied}$ - $N\text{-}NH_{4\ manure}$	Cattle manure, Grassland	Switzerland	7, (5, 6)
Brentrup *et al.*, 2000	Horlacher and Marschner, 1990_b	- $N\text{-}NH_{4\ manure}$ - Operation & incorporation date - Infiltration rate - Temperature, Precipitation	Cattle slurry	Germany	8, 9, (5, 6)
Nemecek and Kägi, 2007	Menzi et al., 1998_b	- $N\text{-}NH_{4\ manure}$ - Fraction $_{area\ spread}$	Cattle manure, Grassland	Switzerland	1
N_2O emissions					
EPA, 1995	Literature review	**EF** % of $N_{fertilizer}$ - Fertilizer type	Fluid fertilizer	US	(11)
Audsley *et al.*, 1997	Amstrong-Brown, 1994_c	**EF** % of $N_{fertilizer}$ - Fertilizer type, Temperature	Granular fertilizer, wet soil	UK	2, 4, 9_e, 7, (3, 5, 6)
Brentrup et al., 2000	IPCC, 1996	**EF= 1.25%** of $N_{fertilizer}$ minus NH_3 and NO_x losses	Mineral and organic fertilizer and manure	Global	8, 10, (3, 5, 6)
Nemecek et al., 2007	IPCC, 1996_b, Schmid et al., 2000_c	**EF= 1.25** % of N_{soil} , **EF= 1.25** % of $N\text{-}NH_3$ losses, **EF= 2.5 %** of $N\text{-}NO_3$ losses, - $N_{available\ fertilizers}$, $N_{crop\ residues}$, N_{fixed}	Mineral and organic fertilizer and manure	Global	(1)
NO_x emissions					
Audsley *et al.*, 1997	No reference	**EF= 10%** of N- N_2O losses	na	na	2, 3, 4, 5, 6, 7, 9, 10
Nemecek and Kägi, 2007	No reference	**EF= 21%** of N_2O losses	na	na	(1)
N_2 emissions					
Brentrup *et al.*, 2000	Rheinbaben, 1990_c	**EF= 9%** of N applied	Arable crop & Grassland	Germany	2, 8
NO_3 emissions					
Audsley *et al.*, 1997	Simmelsgaard, 1998_c	- Reference leaching for recommended $N_{fertilizer}$ - $N_{fertilizer}$	Arable crops & pastures, sandy & loamy soil	Denmark	2, 7, 10_b (3_f , 5, 6)
Brentrup *et al.*, 2000	Own method	- Soil texture, $NO_3\text{-}N_{soil}$ - Precipitation	Crop production	Germany [d]	8, (3, 5 ,6)
Nemecek and Kägi, 2007	Richner et al., 2006_c	- N_{uptake} , $N_{fertilizer}$ - Fraction $_{N\ soil\ leachable}$ - $N_{mineralized}$, Soil depth	Arable crops & meadow	Switzerland	(1)
	Cowell and Clift, 1998	15 kg $N\text{-}NO_3.ha^{-1}$	na	na	9
	Sedilot et al., 2002	**EF= 20%** (soil) and **48%** (substrate) of $N_{applied}$	Greenhouse	France	4

a. Between brackets are uncertain uses (not specific information in papers), **Corresponding papers: 1**. Abeliotis et al. (2013), **2**. Romero-Gámez et al. (2012), **3**. Torrellas et al. (2012), **4**. Boulard et al. (2011), **5**. Martínez-Blanco et al. (2011), **6**. Martínez-Blanco et al. (2010), **7**. Martínez-Blanco et al. (2009), **8**. de Backer et al. (2009), **9**. Mila i Canals et al. (2007 ; 2008), **10**. Antón et al. (2005a); **b**. Adaptation from the original method; **c.** Reference not published in English; **d.** Suggested value for parameters adapted to German climate**, e.** Extended to organic nitrogen with 20% of N as ammonium and 30% as urea; **f.** no nitrate leaching for recirculation irrigation systems; ***na*** = information no data available**.**

1.3.3. Input flows and yields

Table 1.2 presents foreground data related to the field production stage for 72 cropping systems aggregated into 7 cropping system types presenting 7 product groups. For studies comparing systems according to different allocations rules, each system associated with a given allocation ratio was considered original and entered as an additional system.

Data completeness and reference flow

The completeness of field production data varied across studies. Data regarding energy consumption were not provided in 4 studies (Romero-Gámez et al. 2012; Martínez-Blanco et al. 2011; Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2009). Torrellas et al. (2012) and Boulard et al. (2011) did not provide data on fuel consumption for field operations. Martínez-Blanco et al. (2009) did not provide specific data for water consumption.

Table 1.2 presents total fresh matter yields including commercialized and downgraded products. Those yields ranged from 2 $t.ha^{-1}$ for giant bean grown open-field in a Mediterranean context (Abeliotis et al. 2013) to 523 $t.ha^{-1}$ for tomato grown on substrate under heated greenhouse (Torrellas et al. 2012). Tomato showed yields above 100 $t.ha^{-1}$ while the other crops showed yield below 60 $t.ha^{-1}$, regardless of the cropping system type.

Energy and input flows per ton of fresh matter yield

All cropping systems consumed diesel for field operations, except lettuce grown open-field in a tropical context (Mila i Canals et al. 2008). Diesel consumption ranged from 28 $MJ.t^{-1}$ for tomato grown under cold greenhouse on substrate (Antón et al. 2005a) to 1,237 $MJ.t^{-1}$ for giant bean grown open-field in a Mediterranean context (Abeliotis et al. 2013). Some systems were equipped with electrical pumps for irrigation leading to an electricity consumption from 5 $MJ.t^{-1}$ for tomato grown on soil under cold greenhouse (Antón et al. 2005a) to 301 $MJ.t^{-1}$ for giant bean grown open-field in a Mediterranean context (Abeliotis et al. 2013). This latter figure was due to its particularly low yield. Regarding greenhouse heating, the sources of energy were electricity, natural gas or fuel, and in some cases included a co-generation of electricity from fossil fuel leading to negative electricity consumption (Torrellas et al. 2012). The global energy consumption for heating varied from 23,970 $MJ.t^{-1}$ for tomato grown on substrate (Boulard et al. 2011) to 34,170 $MJ.t^{-1}$ for a similar system presented by Torrellas et al. (2012).

All cropping systems were irrigated, except leek (de Backer et al. 2009) and broccoli grown open-field in a temperate context (Mila i Canals et al. 2008). The water volumes applied ranged from 1 $m^3.t^{-1}$ for lettuce grown open-field in a tropical context (Mila i Canals et al. 2008) to 1,570 $m^3.t^{-1}$ for giant bean

grown open-field in a Mediterranean context (Abeliotis et al. 2013). However 80% of systems showed water volumes below 100 $m^3.t^{-1}$. Systems showing the highest water consumption per ton were giant bean, green bean and broccoli grown open-field in Mediterranean context with very low crop yields.

All cropping systems received synthetic fertilizers, except for the lettuce in open-field cropping systems in Uganda (Mila i Canals et al. 2008). Variations of fertilizer inputs across systems were lower than variations for other inputs. Nitrogen rates ranged from 1 $kg.t^{-1}$ for lettuce grown open-field in a tropical context (Mila i Canals et al. 2008) to 19 $kg.t^{-1}$ for broccoli grown open-field in a Mediterranean context (Mila i Canals et al. 2008). Cauliflower and tomato grown open field in Spain (Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2009; Martínez-Blanco et al. 2011), green bean and leek grown open field in UK and Belgium (Mila i Canals et al. 2008; de Backer et al. 2009) did not receive any phosphate-based synthetic fertilizers. Phosphate rates reached 29 $kg.t^{-1}$ for green bean grown open-field in a Mediterranean context (Romero-Gámez et al. 2012). Applications of organic fertilizers were not systematic and never occurred for heated greenhouse types and types on substrate. The types and input rates of organic fertilizers were systematically described. However the nutrient contents of the organic fertilizers were not homogeneously reported in the studies. Organic fertilizers rates reached 4 $t.t^{-1}$ for giant bean grown open-field in a Mediterranean context (Abeliotis et al. 2013).

Table 1.2 Inputs flows for the field production stage expressed per ton of fresh yield and fresh yield for the 72 systems aggregated by cropping system type, product group and reference.

Cropping system type	Product group	Reviewed paper	CS [a]	Fresh yield [b]	Energy			Water	Synthetic fertilizers		Organic fertilizers
					Electricity	*Gaz/Fuel*	*Diesel*		Nitrogen	Phosphate	
				t.ha^{-1}	*MJ.t^{-1}*	*MJ.t^{-1}*	*MJ.t^{-1}*	m^3.t^{-1}	*kg N.t^{-1}*	*kg P_2O_5.t^{-1}*	*t.t^{-1}*
GH-Heat-Sub	**Tomato**	Torrellas et al., 2012	4	523	-2835	37026	na	14	4	2	0
		Boulard et al., 2011	8	440	0	23970	na	29	6	3	0
GH-Heat-soil	**Lettuce**	Mila i Canals et al., 2008	2	58	1153	27498	46	7	1	1	0
GH-Cold-Sub	**Tomato**	Torrellas et al., 2012	1	165	0	0	na	29	5	3	0
		Anton et al., 2005	2	150	5	0	28	24	5	6	0
GH-Cold-Soil	**Green bean**	Romero-Gamez et al., 2012	4	55	na	na	na	88	1	7	0
	Tomato	Boulard et al., 2011	1	150	0	0	na	33	3	2	0
		Martinez-Blanco et al., 2011	3	161	na	na	na	33	2	0	0
		Anton et al., 2005	1	120	5	0	35	29	6	3	0
	Broccoli	Mila i Canals et al., 2008	4	12	0	0	486	0	14	3	0
	Leek	de Backer et al., 2009	2	33	0	0	156	0	1	0	1
	Lettuce	Mila i Canals et al., 2008	6	22	0	0	742	37	6	1	0
OF-Med	**Giant Bean**	Abeliotis et al., 2013	7	2	301	0	1237	1570	10	20	4
	Green bean	Romero-Gamez et al., 2012	2	19	na	na	na	265	4	29	0
	Broccoli	Mila i Canals et al., 2008	4	16	0	0	521	140	19	5	0
	Cauliflower	Martinez-Blanco et al., 2010	3	57	na	na	na	41	3	0	0
	Lettuce	Mila i Canals et al., 2008	4	36	0	0	211	54	10	5	1
	Tomato	Martinez-Blanco et al., 2009	6	124	na	na	na	na	3	0	0
		Martinez-Blanco et al., 2011	3	127	na	na	na	45	3	0	0
OF-Trop	**Green bean**	Mila i Canals et al., 2008	2	24	0	0	42	36	1	1	1
	Lettuce	Mila i Canals et al., 2008	1	41	0	0	0	1	0	0	0

GH = Greenhouse; **OF** = Open-field; **Temp** = Temperate climate; **Med** = Mediterranean Climate; **Trop** = Tropical climate, **na** = data were not available in the published documents **a.** Number of cropping systems aggregated within the line **b.** Fresh yield expressed in t.ha^{-1} for one crop cycle, including downgraded products

1.3.4.LCIA results

Figure 1.1 presents LCIA results expressed per kilogram of fresh matter yield for the three impact categories selected (global warming, acidification and eutrophication potentials) for the 72 cropping systems of the 10 reviewed studies. The 72 cropping systems were successively grouped by cropping system type, product group and reviewed paper. When LCIA results were expressed using functional units involving quality aspects, necessary conversions were used to homogenize the functional unit to yields presented in table 1.2. LCIA results expressed per square meter and crop cycle are given in Annex 1.

Global Warming Potential

The cropping system types under heated greenhouse: GH-Heat-Sub and GH-Heat-Soil had a greater global warming potential (GWP) than other types, at 2.03 and 2.12 kg CO_2-eq. per kg of fresh yield, respectively (Figure 1.1). GWP variations across GH-Heat-sub systems were mostly due to differences in the origin of the energy. Indeed, GWP from GH-Heat-Sub systems presented by Torrellas et al. (2012) showed important variations (± 58%) mainly due to energy sources and associated allocations and substitution rules (gas=5.00 kg CO_2-eq.kg^{-1} fresh yield; co-generation + allocation= 2.00 kg CO_2-eq.kg^{-1} fresh yield; co-generation + avoided burdens= 0.78 kg CO_2-eq.kg^{-1} fresh yield; geothermal= 0.44 kg CO_2-eq.kg^{-1} fresh yield). GWP variations across the GH-Heat-Soil systems were due to different energy consumptions for greenhouse heating. According to authors presenting the two lettuce cropping systems (Mila i Canals et al. 2008), one farm grew only one cycle of lettuce indoors, very likely during the coldest period, while the other farm grew lettuce indoors all year-round, with on average 5 cycles per year. The variation in energy consumption between systems of the GH-Heat-Soil type could result from the crop cycle position within the year. GWP per kg fresh yield for all other types ranged from 0.05 (OF-Trop) to 0.45 kg CO_2-eq. (GH-Cold-Soil) without showing clear differences due to large intra-type variations. For example, the higher impact of green bean under cold greenhouse results from the misting system used to cool down the greenhouse temperature.

Products groups showed undifferentiated GWP values (Figure 1.1), ranging from -0.36 (Broccoli) to 0.89 kg CO_2-eq.kg^{-1} fresh yield (Tomato). Negative GWP was observed in the articles by Martínez-Blanco et al. (2009, 2010 and 2011) due to the avoided dumping of organic wastes in systems using compost as a fertilizer. For example, the tomato system with compost only (Martínez-Blanco et al. 2009) showed a GWP of 0.15 kg CO_2-eq. kg^{-1} fresh yield without considering avoided burdens and -0.90 kg CO_2-eq. kg^{-1} fresh yield with avoided burdens.

While being similar per kg fresh yield, the GWP for GH-Heat-Soil and GH-Heat-Sub systems were significantly different per unit of area (Annex 1), the lowest being for the GH-Heat-Soil type. Such discrepancy between impacts expressed by the two functional units resulted from the combined effect of yield per day and crop duration, the lettuce being grown for two to three months only while tomato

were grown for up to 11 months. Moreover, while showing undifferentiated values between product groups per kg fresh yield, the GWP per unit of area for the tomato group clearly exceeded the GWP of other product groups. The sensitivity of results to the FU choice highlighted here the high intensity of management (quantity of infrastructure and energy used per area) for tomato system grown under heated greenhouse.

Acidification

Regarding the acidification potential (AP), expressed in g of SO_2 equivalent per kilogram of fresh yield, cropping system types did not present clearly different means due to large variations within types and products groups. However, we could relate the ranking of impact between product groups to the fertilizer efficiency, i.e. the ratio of fertilizers inputs per kilogram of fresh yield (as shown in table 1.2). The giant bean group (Abeliotis et al. 2013), presenting the lowest manure efficiency (4t. t^{-1} fresh yield) as well as a low mineral nitrogen efficiency (10 kg $N.t^{-1}$ fresh yield), had the largest impact with an average of 11.26 g SO_2-eq. per kg of fresh yield. The authors of this study identified field emissions from sheep manure as the major contributor for AP. The second greater impact (3.26 g SO_2-eq.kg^{-1} fresh yield) was shown by the broccoli group, which showed the lowest mineral nitrogen efficiency (19 kg $N.t^{-1}$ fresh yield). The particularly low yields shown by these two groups, suggested that AP variations were greatly affected by yields. The lowest impact (0.42 g SO_2-eq.kg^{-1} fresh yield) was shown by the leek group, presenting one of the highest mineral nitrogen efficiency (1 kg $N.t^{-1}$ fresh yield).

The field emissions and low yields of giant bean and broccoli contributed largely to the high AP of the Mediterranean open-field systems type, averaging 4.28 g SO_2-eq. kg^{-1} fresh yield. However, this mean was associated with a particularly large coefficient of variation (± 72 %) stemming from broad variations of input efficiency across systems within this type.

Variations of AP per unit of area (Annex 1) regardless of aggregation mode were greater than variations for AP per kg of vegetable (± 156% vs. ± 71%). Such differences between results expressed by the two functional units highlighted the yield effect, as already suggested previously. The variations of AP between cropping system types and product groups were reduced per kg fresh yield by integrating an aspect of input efficiency, as yields are supposed to be correlated to inputs of systems. Similarly to GWP, the ranking of impact per unit of area for product groups and cropping system types differed from the ranking using the functional unit per kilogram of fresh yield. Indeed, AP expressed per unit of area for tomato systems clearly exceeded the AP of other products groups. Similarly AP expressed per unit of area for GH-Heat-Soil systems clearly exceeded the AP of other cropping systems types. Again, the sensitivity of results to the FU choice highlighted the high intensity of fertilization management (quantity of fertilizer per unit of area) for tomato systems grown under heated greenhouse.

Eutrophication

Regarding the eutrophication potential (EP) expressed in g of PO_4^{3-} equivalent, giant bean systems showed a greater impact (1.75 g PO_4^{3-}-eq.kg^{-1} fresh yield) than other products groups. Broccoli and green bean systems also showed a high impact (1.01 and 0.95 g PO_4^{3-}-eq.kg^{-1} fresh yield respectively). For broccoli systems, such high impact mainly resulted from low mineral nitrogen efficiency, while for green bean it resulted from low mineral phosphorus efficiency (29 kg P_2O_5. t^{-1} fresh yield). For giant bean the high impact resulted from both low phosphorus and nitrogen efficiencies (10 kg N and 20 kg P_2O_5. t^{-1} fresh yield, respectively).We can notice that despite being a leguminous crop (i.e. fixing nitrogen from the atmosphere), the giant bean group presented relatively high nitrogen inputs contributing to its high EP. The Cauliflower group showed less impact than other groups (-1.53 g PO_4^{3-}-eq.kg^{-1} fresh yield) due to the absence of phosphorus inputs, the high mineral nitrogen efficiency (3 kg N. t^{-1} fresh yield) and the system expansion rules used by Martinez-Blanco et al. As already explained for GWP, the latter obtained a negative EP thanks to negative nitrogen and phosphorous emissions from avoided organic waste dumping in the system using compost as a fertilizer. Despite low fertilizer rates, the leek group showed the third greater impact (0.68 g PO_4^{3-}-eq.kg^{-1} fresh yield) due to important phosphorus emissions.

System expansion related to compost fertilization contributed largely to artificially reduce the average EP of Mediterranean open field cropping systems (only 0.45 g PO_4^{3-}-eq.kg^{-1} fresh yield despite the high impact of giant beans) which presented the highest standard error (211%) across all types, products and impact categories. Across all types, GH-Heat-Sub showed the greatest EP (1.08 g PO_4^{3-}-eq.kg^{-1} fresh yield). This greatest impact occurred despite the avoided electricity production thanks to the use of natural gas for co-generation assumed by Torrellas et al. (2012). Over the 12 GH-Heat-Sub systems, one system presented in Torrelas et al. (2012) and 20% of systems from Boulard et al. (2012) were closed-loop.

Similarly to AP, the ranking of impact per unit of area for product groups and cropping system types differed from the ranking of impacts per kilogram of fresh yield: the EP per unit of area of GH-Heat-Soil systems exceeded the EP of other cropping systems types, including GH-Heat-Sub and the EP per unit of area of tomato systems exceeded the EP of other products groups. This latter result highlighted once again the high intensity of fertilization management per area for tomato system grown under heated greenhouse.

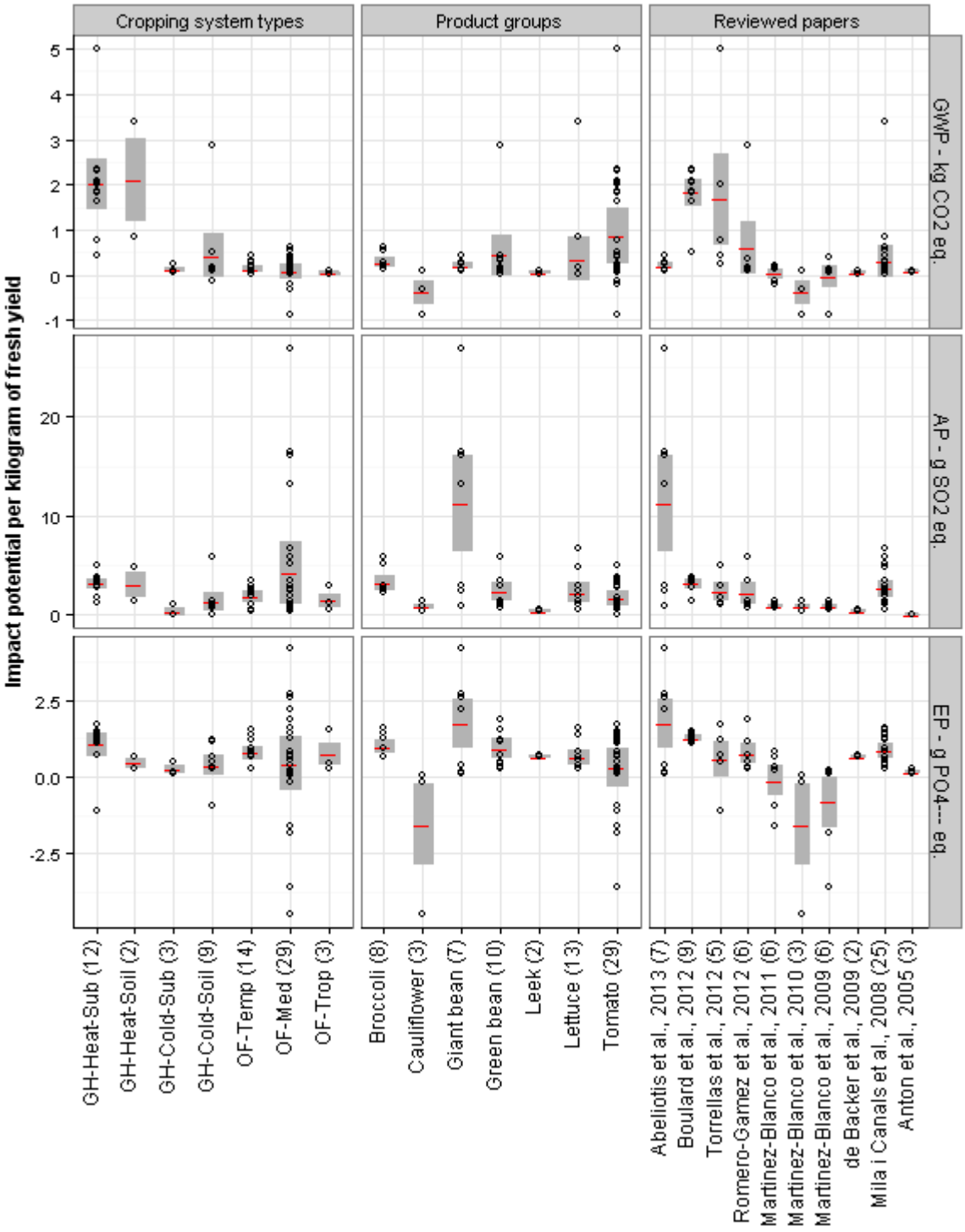


Figure 1.1 Global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP) expressed per kg of fresh yield from cradle-to-harvest for the 72 systems aggregated per cropping system type, product group and reviewed paper. Figures between brackets correspond to the number of systems in the group. Grey boxes correspond to error bars centered on the mean of the group (red dash).

1.4. Relevance of the typology to compare LCIA results from a diversity of cropping systems and vegetable products

While it is difficult to thoroughly analyze LCIA results from several individual studies, using a typology allowed us to provide general conclusions on the environmental impacts of vegetable products and to identify the main cropping systems components responsible for these impacts.

For GWP, we can conclude that the aggregation per cropping system type was the most relevant with heated greenhouse types showing the greatest impact. In agreement with several studies dealing with fruit and vegetable products (Hospido et al., 2009; Stoessel et al., 2012), we identified the type of energy consumed for greenhouse heating as an important source of impact variations. However, GWP for cold greenhouse and open-field types could not be distinguished due to low contribution of infrastructures compared to the impact associated to material and input uses. Our analysis also showed that the crop position within the year led to important GWP variations particularly due to the requirement or not for heating. For AP our analysis suggested that the aggregation per product group was the most comprehensible with the giant bean group showing the greatest impact and the leek group showing the lowest impact. For AP and EP, we identified phosphorus and nitrogen emissions from both mineral and organic fertilizers as the major contributors to the impacts. However, our analysis identified system expansion rules as another important source of variations resulting in the Cauliflower group showing the lowest impact. Finally, comparing two distinct functional units, we showed that LCIA results per kilogram of fresh yield was largely affected by input efficiency per mass of product, while LCIA results per area were largely affected by management intensity per area. This discrepancy led to a change on the ranking of cropping system types and product groups according to the functional unit.

Despite those important insights, the benefits of the typology were hampered by the occurrence of (1) low variations between cropping system types and products and (2) high variations within types and products.

We explored several hypotheses to explain these variations:

1. Inherent variability of cropping systems and products
2. Discrepancies and relevance of methodological choices
3. Limitations in our typology approach
4. We further elaborate on these three potential sources of limitations in sections 1.4.1, 1.4.2 and 1.4.3 with a special focus on the methods used for the estimation of N_r emissions in section 1.4.4.

1.4.1. Inherent variability of cropping systems and representativeness issues

Some variations within cropping system types can be explained by the actual variability attached to cropping systems and products in general. Indeed, the variability of farmer's practices, soil and climate conditions affects cropping systems performances. However, across studies, the poor or unspecified data representativeness reduced the possibilities to identify and interpret this inherent variability.

Across studies, technological, geographical and time-related representativeness of data regarding goal and scope were difficult to assess. Most studies provided detailed information on techniques involved in the field production stage as they were often an object of comparison between systems. However, the time and geographical scales were less detailed. We could locate the data used in time and space but their representativeness regarding the studied region and period was poorly addressed. Only Boulard et al. (2011) provided figures on national production and discussed the representativeness of the data used for the LCA. Abeliotis et al. (2013) tried to address the intra-farms variability proposing a weighted mean resulting from an area-based aggregation of different farms, but did not provide standard deviations. Several LCA studies dealing with vegetable crops (Cellura et al., 2012a) and other crops (Mouron et al., 2006) showed the influence of the management on the environmental performances of systems which pleads for a better inclusion of farms variability in LCA studies.

1.4.2. Discrepancies in methodological choices

System boundaries and completeness of studies

We highlighted a lack of homogeneity regarding system boundaries and cut-off rules. Authors listed and justified most of excluded processes following recommendations from the ILCD handbook (European Commission, 2010). However, the constraint of reporting an LCA through a publication often led to a lack of transparency regarding included processes. In addition several exclusions were not relevant with regard to their potential contribution to the environmental impact of vegetable products.

First, some important stages were not systematically included. *The waste treatment stage* was excluded from 3 studies over the 10 reviewed although Antón et al. (2005b) showed that it contributed for 20 to 94% of GWP and 5 to 50% of EP for tomato production, depending on the type of management considered. Our review did not allow identifying neither the diversity of waste management techniques nor associated impact variations across systems. *The nursery stage* was not explicitly included in 6 studies. To our knowledge there is no literature providing LCIA results for the nursery stage of vegetable. However, for tomato production under cold greenhouse Boulard et al. (2011) identified heating during the nursery stage as the major contributor for GWP of the whole system.

Second, phosphorus emissions were not included in 5 studies with systems fertilized with mineral or organic phosphorus. One of these studies explained that phosphorus was immobilized due to high soil pH resulting in no losses by leaching. However, phosphorus losses occur as soluble phosphate through leaching and run-off but also through the erosion of soil particles. Boulard et al. (2011) showed that phosphate in soil and water contributed from 45% for heated greenhouse on substrate to 63% for cold greenhouse on soil to the eutrophication potential at farm-gate. Due to the heterogeneous description of drainage water management in reviewed papers, we could not analyze eutrophication potential variations related to this aspect. However, Boulard et al. (2011) also stated that the phosphorus emission could probably be reduced by 40% when switching from an open-loop system to a closed-loop system.

Finally, the impacts of pesticide use could not be compared across studies since six of them did not assess human or ecosystem toxicities. According to the ILCD handbook (European Commission, 2010), all impact categories that are relevant for the LCIA study shall be included. Several studies showed that pesticide use dominated the life-cycle toxicity impact of vegetable production (Margni et al., 2002; Anton et al., 2004). Among those papers which included human and ecosystem toxicity impact categories, two did not explicitly include pesticides emissions in the field. Mila i Canals et al. (2008) criticized current estimation methods for being either too simplistic (Audsley et al., 1997) or too impractical (Birkved and Hauschild, 2006), and proposed an additional indicator based on pesticide use.

Transparency and relevance of methods for field emissions

Readers should have the capacity to interpret the contribution of field emissions to the environmental impacts. However, the references used to estimate these fluxes were not sufficiently detailed, neither were the parameters values used for calculation. Four papers did not report the methodology used to estimate each emission from fertilizer and pesticide use. In addition, authors did not detail the parameter values used for the calculations, preventing the reader from recalculating emissions when they were not included in the article. The acidification potential and the eutrophication potential should be largely affected by reactive nitrogen-based and phosphorus-based field emissions. Unlike Boulard et al. (2011) and Antón et al. (2005a), our review did not identify a reduction of the eutrophication potential between systems on substrate and systems on soil, regardless of the functional unit used. In addition we could not properly identify the respective contributions of phosphorus and nitrogen emissions to the eutrophication potential. Considering the particularly weak description of methods to estimate phosphorus emissions and the lack of consensus, we can assume that important EP variations were related to methodological choices.

Detailed information on the estimation methods would have permitted to explain why those systems so diverse a priori showed similar impacts for AP and EP and to assess the relevance of the methods used. This is especially likely for water-borne emissions (the main contributors to EP), since differences in climate, irrigation, fertilizer managements and growing media result in particular water and nutrients dynamics. Since N_r emissions were systematically included in inventories and often responsible for significant acidification and eutrophication potential impacts, a special focus on their estimation methods is proposed in section 1.4.4.

1.4.3. Weaknesses in reporting, a limitation to our typology approach

Our typology was an a priori typology based on a restricted number of characteristics of vegetable cropping systems. A quantitative typology based on a statistical approach (principal component analysis and hierarchical clustering) would have been preferable. However, it was not feasible in our situation given the lack of transparency in the reporting of key cropping systems characteristics across studies. Below, we identify often missing although critical parameters to be included and reported in LCA studies for vegetable products. These parameters could allow improving the relevance of further attempts at characterizing the variability of LCIA results across vegetable products. Such characterization based on a meta-analysis should therefore provide information about important influencing factors for the environmental performances of vegetable products for the attention of policy makers, food industries, consumers and producers.

Our analysis showed that the crop position within the year led to important GWP variations. For some vegetable crops, several cropping cycles can occur on the same field during one year. Cellura et al. (2012a) showed that crop rotations can lead to a clear reduction of all environmental impacts for both Tomato and Melon, especially with respect to waste production and allocation rules.

Our analysis also showed the important contribution of organic fertilizers to AP and EP. However, information regarding the types and composition of organic fertilizers was not homogeneous across studies. The type and the composition of the organic fertilizers inform not only on the nutrient inputs to cropping systems, but also on the environmental risks associated to their application. Indeed, in addition to nitrogen and phosphorus contents, compost (Shiralipour et al., 1992; Weber et al., 2007) and manures are known to be sources of a diversity of trace contaminants such as heavy metals and persistent organic pollutants. Finally, the method of application plays an important role regarding the acidification and eutrophication potentials with some application techniques of manure especially designed to abate ammonia emissions (Langevin, 2010).

Finally, irrigation practices appeared in recent studies in Mediterranean areas as an important contributor for GWP and water consumption (Abeliotis et al. 2013, Cellura et al., 2012a&b, Romero-Gámez et al., 2012). Our analysis did not allow identifying the contribution of irrigation practices to

the impact of vegetable production due to heterogeneous description of energy and water consumed. However, the huge diversity of irrigation practices from recirculation systems to sprinkler irrigation systems should have affected the environmental performances of vegetable production systems.

1.4.4. Relevance and validity of the methods used for the estimation of reactive nitrogen emissions

Field emissions are substances emitted during the field production stage. Emissions can be either directly caused by an input application (fertilizers or pesticides) or by processes occurring in the soil and in the atmosphere (nitrogen deposition, drainage, soil mineralization). Field emissions do not include emissions from fuel burned in tractors. In this section, we focused on reactive nitrogen (N_r) emissions because they were systematically included in inventories and often responsible for significant acidification and eutrophication potential impacts. In table 1.1, the references detailing calculation methods were analyzed regarding the parameters included and the validity domain of the specific method, be it an emission factor or a more complex equation. In the next sub-sections, an in-depth analysis for each emission process is proposed regarding the adequacy of each reference to the specific conditions of the diverse vegetable cropping system types. Recommendations are made in the last section of this paper.

Volatilization of applied fertilizers and associated ammonia emissions

Volatilization is a physico-chemical process leading to ammonia (NH_3) emissions, which have high acidifying properties. In the three guidelines used, NH_3 emissions from mineral and organic fertilizers were dealt with separately.

For mineral fertilizers, the guidelines of Audsley et al. (1997) and Nemecek and Kägi (2007) recommended emission factors from Asman (1992). This reference proposed emission factors measured under controlled conditions in the laboratory for each type of fertilizer used in Europe. These emission factors were used as references for the whole Europe, until the publication of the ECETOC report (1994) recommended by Brentrup et al. (2000). ECETOC (1994)recommended a new set of emission factors based on an extensive literature review, depending on the country considered and the occurrence of soils with a pH greater than 7 (indicative of calcareous soils).

For NH_3 emissions from organic fertilizers, the guidelines from Audsley et al. (1997) and Nemecek and Kägi (2007) recommended a calculation method developed by Menzi et al.(1998). This method was based on experiments in Switzerland, where liquid manure was applied to grassland. Audsley et al. (1997) suggested simplified emission factors depending on the ammonia content of the organic fertilizer. By contrast, Nemecek and Kägi (2007) slightly adapted the empirical equation from Menzi et al. (1998) with parameters related to techniques of application. Finally, Brentrup et al. (2000) recommended an empirical equation developed by Horlacher and Marschner (1990) in a temperate

context for cattle slurry spread on arable crops. Some parameters of this empirical equation may be adjusted to account for different soils, crops and times of application with regard to incorporation or precipitation.

None of those references are actually suitable to estimate emissions from open-field systems under Mediterranean or tropical climates since high temperatures may entail large variations in the emissions, especially if the fertilizer application occurred in the hottest season. For greenhouse systems, the use of those calculation methods was very remote from their validity domain as wind speed and temperature effects were not included, not to mention the effect of plastic mulch. Ammonia emissions are largely variable according to crop type, local context and application techniques (Bussink & Oenema, 1998; Langevin, 2010).

Denitrification of soil nitrogen and associated gaseous emissions

Nitrification and denitrification of soil nitrogen are responsible for the emission of nitrous oxide (N_2O), nitrogen oxides (NO_x) and nitrogen gas (N_2) emissions. N_2O is a potent greenhouse gas, while NO_x contributes mainly to photochemical oxidant formation and terrestrial acidification. In the two guidelines (Audsley et al., 1997; Nemecek and Kägi, 2007) and the original paper from Brentrup et al. (2000), each emission is dealt with separately.

For N_2O emissions, Audsley et al. (1997) associated N_2O emissions to nitrogen from fertilizer application and emission factors were based on an unpublished reference. These emission factors depended on the fertilizer type and the soil temperature and belonged to the range [0.4-3%] (as a fraction of applied N). Later on, the guidelines from Brentrup et al. (2000) and Nemecek and Kägi (2007) recommended emission factors from IPCC (1996), based on an extended review of published field experiments mainly from temperate regions. Brentrup et al. (2000) recommended using the direct emission factor from the IPCC methodology. The direct emission factor corresponds to the proportion of soil nitrogen turnover emitted as N_2O, including N mineralized from mineral and organic fertilizer N inputs, N fixation from legumes, and N in plant residues. In addition to the emission factor for direct emissions, Nemecek and Kägi (2007) also included the two indirect emission factors from IPCC (1996) via the equation developed by Schmid et al. (2000). The indirect emission factors correspond respectively to the fraction of N inputs volatilized as ammonia, which is further deposited to soils and emitted as N_2O and, the fraction of N inputs leached as nitrates and further emitted as N_2O.

Only two guidelines included the estimation of NO_x emissions. Brentrup et al. (2000) neglected this emission, probably because there was no consensus on its emission factor. The guidelines from Audsley et al. (1997) and Nemecek and Kägi (2007) agreed on NO_x emissions equal to 21% N_2O

emissions. None of them quoted any published reference to justify such figure, which is actually contradicted by recent literature (Stehfest & Bouwman, 2006). Finally, although not of environmental relevance, N_2 emissions were included by Brentrup et al. (2000) in their nitrogen balance approach to infer nitrate leaching. They proposed a constant emission factor of 9% of the nitrogen applied according to von Rheinbaben (1990).

Providing a unique emission factor, the IPCC guidelines (Tier 1) do not allow for discriminating between systems but for calculating the soil nitrogen input. Nevertheless, the broad validity domain of this reference and its wide use make it the most appropriate method currently available for open-field systems, even under tropical conditions as a few field experiments used to calculate the emission factors were led in tropical areas. However, its use for greenhouse systems was far beyond the scope of the IPCC report. It does not appear that significant improvements can be possible using a unique emission factor, since many parameters influencing N_2O and NO_x emissions were recently evidenced and their interactions are not fully understood (Stehfest and Bouwman 2006). The effect of temperature and fertilization on soil nitrogen content and soil pH and the effect of the irrigation on soil humidity are particularly relevant to consider for vegetable cropping systems.

Leaching and associated nitrate emissions

Nitrate leaching occurs when there is an accumulation of NO_3 in the soil profile that coincides with or is followed by a period of drainage. Therefore, excessive nitrogen fertilizer or N applications before drainage events, ploughing, or long fallow without cover, can all potentially lead to NO_3 leaching losses. In each guideline analyzed, a specific approach for nitrate leaching estimate is recommended in relation to the specific context where the method was developed.

Audsley et al. (1997) recommended the use of a Danish approach adapted from a statistical equation from Simmelsgaard and Djurhuus (1998). In this approach, a reference leaching loss for a recommended fertilizer rate was used as a parameter to reflect the soil capacity for leaching. The validity domain of this statistical model is restricted to Danish conditions where references are available regarding the reference leaching. Brentrup et al. (2000) suggested the use of a conceptual model based on a nitrogen balance coupled with an empirical equation reflecting drainage events. Although following a more conceptual approach, this equation was calibrated under German conditions for arable crops only. Finally, Nemecek and Kägi (2007) recommended the use of a conceptual approach including a set of emission factors. These emission factors were calculated using a mechanistic model developed by Richner et al.(2006), but were further calibrated under Swiss conditions for arable crops. This conceptual approach included parameters related to soil, plant uptake, soil coverage and risk of nitrogen leaching depending on the crop types and the months in which

fertilizers were applied. This range of parameters allowed the user to adapt the calculation within the range of situations defined in the method (arable crops under temperate climate).

The current methods used to estimate nitrate emissions were more complete in terms of driving parameters compared to other emissions, but remained mostly valid for arable crops in temperate contexts. Overall, the only reviewed study using a suitable method for the estimation of leaching was de Backer et al. (2009) who assessed leek cropping systems (without irrigation) under temperate climate using the Brentrup's method (Brentrup et al. 2000). Otherwise, none of these methods were really suitable for estimating nitrate leaching for any other vegetable cropping systems.

1.5. Conclusions and recommandations

This paper reported on the current practice for the LCA of vegetable products with a focus on the field production stage. It was a first attempt at comparing potential impacts of a large range of cropping systems representative of the current European supply mix of vegetable products. It presented aggregated LCIA results: i) according to the cropping system type through an original typology, ii) according to the product group and iii) according to the reference. Setting a typology of systems appeared as a suitable approach that should be generalized to characterize systems diversity and obtain representative Life-Cycle Inventory (LCI) data, in relation to the function and scale defined in the goal and scope. We can conclude that the heated greenhouse system types showed the greatest GWP impact due to the crop position within the year in relation to energy requirement for heating. When GWP was expressed per area, the tomato group showed the highest impact due to high management intensity per unit of area. Regarding AP and EP the giant bean group showed the greatest impact due to low yields, responsible for low input efficiency. When AP and EP were expressed per unit of area, the tomato group showed the greatest impact due to high fertilizer management intensity per area.

We highlighted important sources of impact variations across systems. Important GWP and EP variations were caused by system expansion rules related to energy and compost used. Important GWP variations were due to the crop position within the year and the crop duration, in relation to energy requirement for heating. Important AP and EP variations were caused by yield variations and diverse fertilization managements.

However, the ability to compare the environmental impact for these diverse field production stages was hampered by (1) weaknesses regarding transparency of goal and scope, (2) a lack of representativeness and completeness of data used for the field stage and (3) heterogeneous and inadequate methods for estimating on field emissions across reviewed studies.

Table 1.3 Summary of recommendations related to key parameters to be included in LCI and methods to be used for the estimation of reactive nitrogen (N_r) emissions to account for specificities of vegetable cropping systems

Specificities of cropping systems	**Key parameters**	**Methods for the estimation of N_r emissions**			
		NH_3 mineral fertilizers	*NH_4 organic fertilizers*	*N_2O and NO_x*	*NO_3*
Vegetable cropping systems	· Fresh matter yield and commercial yield if suitable · Nitrogen and phosphorus inputs along with fertilizer types · Water inputs and origin · Irrigation system specificities · Crop duration and position within the year · Previous and following crop · Crop residue management	×	×	×	×
1. Greenhouse systems	· Greenhouse materials and associated waste management · Allocation rules for infrastructures	ø	ø	ø	Brentrup et al. (2000) adapted to soil type and irrigation management
1.1. Heated greenhouse systems	· Energy inputs and origin · Allocation and substitution rules associated to co-generation				
1.2. Systems with substrate	· Substrate type and associated waste management · Drainage water management	ø		ø	Bres (2009)
2. Open-field systems	· Soil type · Rainfall · Origin and composition of organic fertilizers · Allocation rules associated to organic fertilizers	ECETOC (1994)…	Brentrup et al. (2000) parameterized for…	IPCC (2006) & Nemecek et al. (2007)	IPCC (2006) or Brentrup et al. (2000) adapted to…
2.1. Temperate context		… Gp. I	… soil type and plant cover	○	… soil type and irrigation management
2.2. Mediterranean context		… Gp. III	… soil type and plant cover	○	… soil type and irrigation management
2.3. Tropical context		Bouwman and Van Der Hoek (1997) & ECETOC (1994) Gp. II		○	… soil type, irrigation management and rainfalls

× Level of specificity not homogenous enough to consider only one method, ø level of specificity which requires a specific methods but for which no method is available to date, ○ level of specificity for which a specific methods could allow reducing the uncertainty.

Generally speaking, several aspects would deserve a systematic inclusion and a dedicated reporting in the LCA of vegetable products. First, human and ecosystem toxicity impacts should be more systematically assessed to consider pesticide practices, as well as the impact of freshwater use to account for the frequent use of irrigation and the location of systems often in contexts of water stress. Their absence can be partly explained by the lack of a consensual method at the time of their publication regarding both methods for estimating emissions (for pesticides) and for characterization. For these categories, there is an urgent need for further investigation. Second, geographical and time-scale representativeness should be better addressed with regard to inventory data for the field production stage of vegetable products. In agreement with the ILCD handbook (European Commission, 2010), we strongly recommend providing data describing the sample of farms surveyed with regard to its representativeness in connection with the scope of the LCA. Variability of the management between farms and years should be better accounted for through valid sampling procedures and discussed in the interpretation step, based on consensual quality criteria such as the Data Quality Criteria proposed by Weidema and Wesnæs (1996). Finally, the waste treatment and the nursery stage should be systematically included in inventories with detailed information about techniques involved, inputs and energy consumption and emissions at least until their insignificant contribution would be clearly demonstrated.

More specifically, an effort should be made to collect and report accurate, precise, and complete inventory data regarding the main discriminating characteristics of cropping systems representative for the studied function. Authors should more often take opportunities offered by many journals to provide full inventories in supplementary materials. A quality reporting of key cropping systems' data would allow better quality meta-analyses including statistical analysis as an extension the analysis proposed in this paper. Table 1.3 summarized those key parameters to be included in inventories according to cropping system specificities. In agreement with recommendations from several authors of LCA for vegetable products (de Backer et al. 2009; Andersson et al. 1998), land occupation characteristics (the crop duration, its position within the year, the previous and following crops, crop residues management) along with detailed allocation rules should be described to properly assess the impact of vegetable crops on a yearly-basis. As proposed by several authors, another solution could be to expand the system to the complete rotation. The organic fertilizer management should be thoroughly taken into account for open-field cropping systems, in particular regarding the allocation rules related to its use. There is also a pressing need to investigate impacts related to the contaminants contained in organic fertilizers. Being an important source of nutrients especially regarding integrated and organic production orientations, more detailed data on organic fertilization is deemed necessary. Finally, regarding irrigation practices, the infrastructure and energy consumption should be better documented with regard to its potential contribution to global warming. There is also a pressing need to specify volumes and origin of the water used especially in a context where water is a scarce

resource. The irrigation infrastructure (drip or sprinkler), the quantities and nature of energy used (diesel or electricity), the quantities and origin of water used (river, groundwater) should be systematically detailed in LCA dealing with vegetable products.

Regarding field emissions, we recommend to systematically estimate emissions of phosphorus and pesticides in line with the specificities of the cropping systems evaluated and to report on the methods used for their estimation. Regarding nitrogen reactive emissions, we examined the relevance of the methods used to the diversity of cropping systems studied. Table 1.3 summarizes identified gaps in terms of methods availability and provides recommendations on the best available methods according to cropping system specificities. We believe that available methods could not allow a proper estimation of N_r emissions for all cropping systems, due to the diversity of vegetable systems. For gaseous emissions from systems under greenhouse, there is currently no available method accounting for such confined atmospheres (radiation, humidity and wind), plastic mulching or substrate conditions. Conversely, for open-field systems, we can recommend the use of ammonia emission factors from ECETOC (1994) for mineral fertilizers. For ammonia emissions from organic fertilizers, the method proposed by Brentrup et al. (2000) has the advantage to allow the parameterization of soil type and plant cover. For ammonia emissions under tropical climate, the emission factors proposed by Bouwman and Van Der Hoek (1997) seem the most appropriate, and ECETOC (1994) group III can be used as a complement for fertilizers not dealt with in the article. For nitrous oxide emissions, direct and indirect emission factors proposed by IPCC (2006) seem the most appropriate. For both ammonia and nitrous oxide emissions, further developments are required to calibrate more mechanistic tools easily transposable in Mediterranean and tropical contexts. These tools should at least include parameters to account for temperature, soil type and application techniques. For the estimation of nitrate leaching, we can recommend the use of the nitrogen balance proposed by Brentrup et al. (2000). However, a proper water balance should be done, accounting for the soil and climate specificities in Mediterranean or tropical contexts, especially considering the common intensive practices of irrigation in vegetable systems. For on-substrate systems, the methodology developed by Bres (2009) seems the most appropriate provided that specificities of waste water management are included (closed and open-loop systems). For each type of N_r field emissions, there is a potential for improvement from using measured data in different contexts to adapting empirical equations, or using mechanistic and dynamic models to develop generic tools. The soil nitrogen balance should be expanded to include the dynamics of organic nitrogen from soil, fertilizers and amendments. Indeed, soil of vegetable gardens often shows high organic matter content due to frequent and important organic matter inputs. However, difficulties will arise for short crop cycles and particularly in tropical contexts given the quick turn-over of nitrogen fluxes.

We identified important potentials for improvement in capturing the variability of cropping systems when assessing the environmental impact of vegetable products through the LCA methodology.

However, there will be a trade-off between the additional work required to follow these recommendations and the benefits in terms of results quality, to be balanced according to users' own interest. One important limitation remains the lack of quantitative data in certain contexts, such as developing countries. Data availability has to be included as a constraint for the development of methods, and different but consistent tiers of estimation tools should be proposed to users, similarly to the IPCC guidelines for GHG inventories.

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Contribution du chapitre 1 et perspectives dans le cadre de la thèse

Les principaux résultats du chapitre sont résumé en figure 1.2. Dans le cadre de la thèse, ce chapitre nous a permis de mieux définir les objectifs et le cadre de notre étude. Nous avons ainsi identifié 2 unités fonctionnelles pertinentes et complémentaires : un kilogramme de produit et un hectare de production. Nous avons également identifié le périmètre des champs cultivés comme un cadre pertinent pour une analyse approfondie de la diversité des systèmes de production et de la variabilité des émissions azotées.

La quantité et la qualité des données nécessaires pour la réalisation des inventaires apparaissent comme un verrou majeur pour l'application de l'ACV aux produits maraichers, soulignant la nécessité d'une collecte de données de qualité pour notre cas d'étude que nous mettrons en œuvre dans le cadre du chapitre 2. Nous avons identifié la typologie comme un moyen de caractériser la diversité des systèmes de production avec des perspectives intéressantes dans le cadre de l'ACV.

Nous avons également identifié les limites des méthodes d'estimation des émissions azotées couramment utilisées en ACV. La modélisation biophysique est une piste prometteuse pour raffiner l'estimation des émissions que nous explorerons dans le chapitre 3.

Enfin, nous avons mis en évidence l'influence des choix méthodologiques sur les résultats d'ACV (limites du système, allocations), et l'importance de l'aspect multicritères et de l'analyse de sensibilité pour produire des références fiables et transparentes.

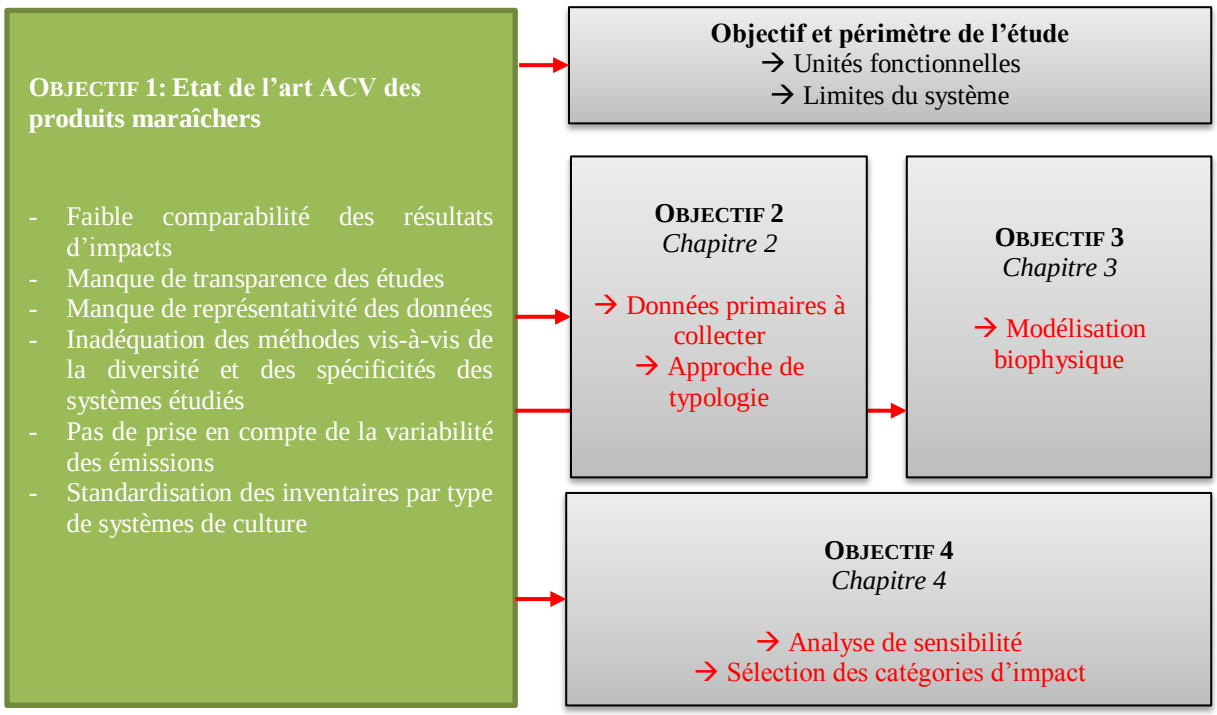


Figure 1.2 Résultats du chapitre 1 (boite verte) et contributions pour les chapitres suivants (rouge)

Chapitre 2. Méthodes et données pour une analyse du cycle de vie fiable et représentative dans un contexte de faible disponibilité en données

Le cas de la tomate en maraîchage urbain au Bénin



Traitement phytosanitaire dans un jardin urbain à Cotonou, Bénin

Avant-propos

Comme souligné en introduction, le premier défi à relever dans le cadre de cette thèse est la création de données représentatives pour l'évaluation environnementale de la tomate en maraîchage urbain au Bénin. En effet, contrairement aux grandes cultures dans les pays développés, le maraichage urbain est une production encore mal connue et peu décrite. Dans le chapitre 2 nous avons identifié l'influence de la diversité des systèmes de production sur les résultats d'ACV. Au préalable de l'évaluation environnementale de notre système, il nous a donc semblé nécessaire de mieux comprendre les systèmes de culture de tomate en jardin urbain au Bénin, leurs spécificités et leurs performances agronomiques. C'est la démarche que nous proposons dans ce chapitre. Ce passage sur le terrain est loin d'être systématique dans la pratique de l'ACV. Nous montrerons qu'il est pourtant nécessaire surtout dans des contextes d'agriculture tropicale où pratiquement aucune donnée n'est disponible dans des bases de données ou des rapports.

Cette étude s'est déroulée sur la zone côtière du Bénin. Dans le sud du pays, le climat est de type équatorial avec 2 saisons sèches et deux saisons humides. La tomate que nous avons étudiée est produite pendant la longue saison sèche qui s'étend de Septembre à Avril. Nous avons étudié une saison de production à cheval sur les années 2011 et 2012.

Nous nous sommes plus particulièrement intéressés aux jardins urbains localisés aux alentours et au cœur de 3 grandes villes côtières: Cotonou, la capitale économique, Ouidah et Grand-Popo (Figure 2.1). Nous avons choisi ces zones de production car elles concentrent les plus hauts niveaux de densité de population.

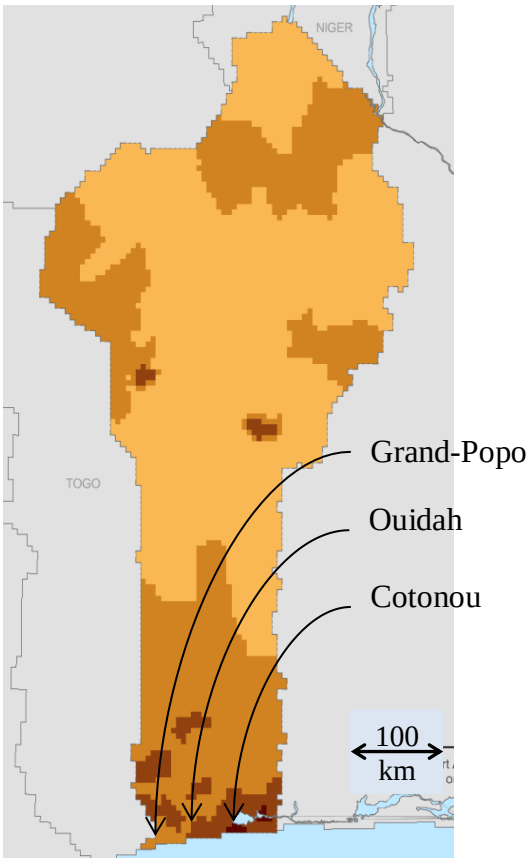


Figure 2.1 Densité de population au Bénin (Center for International Earth Science Information Network)

Dans ces jardins urbains on trouve des sols principalement sableux sauf le long des rivières et lagons ou les teneurs en limon et argiles peuvent être importantes. Ces systèmes sont caractérisés par un usage systématique d'engrais et de pesticides. Nous avons suivi l'ensemble des opérations culturales pendant environ 100 jours pour chaque parcelle du stade pépinière à la dernière récolte. Au total cela représente 6 mois de travail de terrain réalisé avec l'aide précieuse d'un intermédiaire sur place (chercheur Cirad expatrié) et d'un stagiaire Béninois.

Résumé

En Afrique sub-saharienne, l'agriculture urbaine présente un fort potentiel de réduction de la pauvreté urbaine et d'amélioration de la sécurité alimentaire. Pour faire face à la demande croissante des villes en produits frais et à la pression foncière, les maraîchers ont intensifié leurs systèmes de production. Ainsi l'augmentation de l'utilisation d'engrais et de pesticides induit des risques croissants sur l'environnement et la santé humaine. Afin d'encourager le développement de systèmes maraîchers urbain et périurbains durables en Afrique, il est capital d'avoir des données fiables sur ces systèmes (ex. rendement, engrais, pesticides, eau). Ces données doivent être représentatives de la diversité des pratiques afin de permettre une évaluation environnementale pertinente. En se basant sur l'étude de la tomate de contre-saison dans les jardins urbains au Bénin (Afrique de l'Ouest), l'objectif de cet article est d'une part de fournir une méthode permettant de réaliser des Inventaires de Cycle de Vie de bonne qualité, dans un contexte où les données sont rares, et d'autre part de fournir de nouvelles données agronomiques pour les systèmes maraîchers urbains d'Afrique.

Une procédure originale basée sur la typologie a été développée et a permis la production de données d'inventaire de qualité, selon l'index de qualité des données recommandé en ACV. Les ordres de grandeur obtenus pour les paramètres clés tels que le rendement, l'eau, l'utilisation des engrais et des pesticides et les émissions aux champs ont été validés par comparaison avec des données mesurées disponibles dans la littérature. Notre étude démontre pour la période et l'échantillon étudiés que les pratiques agronomiques ne sont pas optimisées, avec une forte variabilité des rendements et une utilisation fluctuante et souvent excessive d'engrais et de pesticides. Au regard de ces données d'inventaire, l'évaluation environnementale de ces produits par l'ACV devrait inclure au minimum : l'écotoxicité pour les milieux aquatiques et terrestres, l'eutrophisation, l'acidification, le changement climatique et la consommation d'eau. Enfin, ce travail montre qu'un important effort de recherche est encore nécessaire afin de mieux connaitre le devenir des nutriments et des pesticides dans les systèmes de maraîchage urbain Africains.

Mots-clefs : Agriculture urbaine ·Analyse du Cycle de Vie · Typologie · Création de données ·Diversité · Scénario

Life Cycle Inventory for tomato urban gardens in Benin: low performances and high environmental risks systems

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Abstract

To face the increasing demand for fresh products from sub-Saharan African cities and the scarcity of land, urban vegetable growers intensified their practices leading to potential risks for the environment and for human health. Comprehensive environmental assessments, representative for a diversity of agricultural practices are required and should be based on consensual methodology such as Life Cycle Assessment (LCA). To feed the life cycle inventory (LCI) on which LCA is based, data on production systems (pesticide use, fertilizer use and field emissions, water use and yield) is needed. Based on the case study of out-of-season tomato from Beninese urban gardens, the objectives of this paper were (1) to evaluate the performances and the environmental risks associated to tomato production in Benin and (2) to assess the quality of LCI data using an LCA-dedicated procedure: the Data Quality Index.

An expert-based typology was used to select among 40 farmers, 12 cropping systems to be monitored for six months and build representative LCI data to be compared with the literature. Treatment Frequency Index (TFI) nutrient budgets and field emissions, were calculated based on best available methods.

We presented LCI data for the main cropping system types and their weighted mean. Pesticide and fertilizer rates were variable and excessive. The weighted mean TFI for insecticide reached 8.9. Beyond the large field emissions to the environment, the weighted mean nutrient budget showed large nutrient surplus: 120 kg N and 84 kg P. ha^{-1}. Yields were low (11,501 $t.ha^{-1}$ for weighted mean) and also variable. Overall, practices were not optimized and revealed low performance and high risks for the environment. According to the data quality index, the LCI dataset was of high quality, enabling a future environmental assessment with the LCA methodology, which should at least include ecotoxicity, eutrophication, acidification, global warming and water deprivation potentials.

Keywords Urban agriculture · Life-cycle assessment · Typology · Cropping system data · Treatment frequency index · Nutrient budget

2.1. Introduction

In sub-Saharan Africa, the urban population is projected to increase from 400 million to 750 million between 2010 and 2030 (UN-HABITAT, 2013). Urban agriculture holds the most potential to feed this increasing population, curb urban poverty and ensure food security (Abdulkadir et al., 2012). To face the increasing demand of fresh products from the cities and the scarcity of land available for agriculture, urban farmers and especially vegetable gardeners intensified their practices (multiple crop cycles in one year, continued cropping for many years) leading to improper uses of fertilizers and pesticides (Drechsel & Zimmermann, 2005; De Bon et al., 2010). Such intensive practices started to lead to deleterious environmental consequences (De Bon et al., 2010; Abdulkadir et al., 2012). In this concerning context, a comprehensive environmental assessment, representative for a diversity of agricultural practices for urban horticulture in Africa is required.

Comprehensive environmental assessments for agricultural products have been successfully based on Life Cycle Assessment (LCA) which is a multicriteria and global approach. LCA is a normalized methodology (ISO 14040, 2006; ISO 14044, 2006) based on a comprehensive environmental inventory, or Life Cycle Inventory (LCI), which compiles all resources that are needed for and all emissions that are released by the specific system under investigation and relates them to the function of the system. However, this powerful decision helping tool has almost never been used to assess the environmental impacts of vegetables in tropical contexts (Sim et al., 2007; Mila i Canals et al., 2008). Applying the LCA framework to tropical vegetables constitutes the most challenging task for practitioners because of the considerable variations in the resource use and performances between farms of the same main activity (Haas et al., 2000) and also because of the extreme scarcity of data in such contexts.

By quantifying all inputs and outputs including emissions to the environment, the LCI step of the LCA methodology offers a consistent framework to study the performance and environmental risks of urban horticulture systems. Indeed, emissions to the environment constitute risks, or hazards, for receiving ecosystems. However to date, the literature on African urban horticulture does not offer sufficient data including quantified inputs and outputs as required for a proper life cycle inventory. Although several studies addressed environmental issues of urban horticulture in Africa, they focused either on fertilizers or pesticides. Regarding pesticides, studies suggested frequent treatments with large rates of active substances (Rosendahl et al., 2009; Ahouangninou et al., 2012). However to our knowledge, no paper provided quantified and comprehensive data on crop protection practices for urban cropping systems in Africa. Regarding fertilizers, large quantities of nutrients in the form of chemical fertilizers and livestock manure are applied to urban gardens (Diogo et al., 2010; Sangare et al., 2012). Due to low yields, an important part of nutrient input is lost to the environment (Predotova et al., 2010; Predotova et al., 2011; Lompo et al., 2012). Nutrient emissions (as NH_3, N_2O, NO_3 and phosphorus)

often contribute considerably to the environmental risk associated to urban cropping systems. However, it is often difficult to estimate rates of nitrogen and phosphorus released to the air and to the water, as measuring emission rates directly on field is time and cost-demanding, especially when considering the variability of emissions at small scale (Bussink & Oenema, 1998; Stehfest & Bouwman, 2006).

Furthermore, recent LCA guidelines have emphasized the importance of evaluating the quality of LCI data in terms of reliability, completeness and representativeness of the studied function (ILCD handbook, European Commission, JRC, 2010). An LCA-dedicated procedure is based on the so-called pedigree matrix including the five quality criteria proposed by Weidema and Wesnæs (1996): Reliability, Completeness, Temporal correlation, Geographical correlation and Technological correlation. Although recent papers showed interesting attempts to account for the diversity between horticulture farms (Cellura et al., 2012; Ingwersen, 2012), most LCA studies still lack a comprehensive discussion of the representativeness and quality of LCI data and this recommended procedure appears insufficiently applied. This is particularly detrimental considering the wide diversity and variability of those systems and the tremendous lack of data.

Based on a comprehensive Life Cycle Inventory procedure applied to a typical African urban garden case study showing extreme data scarcity, the objectives of this paper were (1) to evaluate the performances and the environmental risks associated to tomato production in Benin including cropping systems diversity and one representative scenario, and (2) to evaluate and discuss the quality of LCI data according to recommended LCA procedure.

2.2. Materials and methods

2.2.1. Study sites and characterization of cropping systems

Our data collection took place in Southern Benin from September 2011 to April 2012 which corresponds to the long dry season (tropical climate). In this region, urban gardens are essentially located within or in the surrounding of three coastal cities, Cotonou, Ouidah, and Grand-Popo, thanks to the accessibility to water for irrigation and the proximity of urban markets. Those urban gardens present sandy soils except for those located along the rivers or lagoons where the soil can content high rate of silt and clay. Among vegetable productions, the out-of-season tomato is one of the highest value crops grown by urban farmers, and aims at providing cash flow for urban farmers. Indeed the tomato being the base for many traditional dishes, the important demand leads to high prices on the market during this unfavorable period for tomato production (low rainfall and high pest pressure). Unlike the seasonal tomato production mostly located in rural areas, the out-of-season production requires irrigation. In addition, large amount of fertilizers and pesticides are applied to increase yields

and cope with the high land pressure in urban area. Those cropping system specificities are of major importance for the environmental performance of tomato production in urban gardens in Benin.

With the support of the communal extension services of the three cities, we recorded 40 urban farmers growing out-of-season tomatoes during the 2011-2012 season (see Annex 2). We implemented a typology of the out-of-season tomato cropping systems— a cropping system being defined as a crop grown with a specific soil, climate, and management. This typology addressed a dual-purpose: (1) to allow the sampling of representative cropping systems for the population to which applying a detailed monitoring (section 2.2.2), (2) to quantify the representativeness of the data and allow the production of a unique representative scenario (section 2.2.3). To this aim, we identified three criteria easily observable and quantifiable on farms—type of irrigation system; type of water resource; and planting date—with the help of a panel of Tropical horticultural experts from CIRAD (the French research center for agriculture in developing countries), from INRAB (the Beninese national research institute of agriculture) and from the governmental extension service (CeRPA) which was identified as the best mean to access farmers. For each criteria, three modalities were also identified: (1) type of irrigation system: manual, hose, and sprinkler (being the most advanced system); (2) type of water resource: depression (the most restricted resource), drilling, and river; (3) planting date: early (September), median (November), and late (January).

2.2.2. Sampling cropping systems for monitoring

To comply with the need for a good representation of the tomato cropping systems diversity identified through our typology and the time required for a quality monitoring, the sampling consisted in the selection of 12 cropping systems. Indeed, to capture the main changes in daily cultural operations and to record precisely occasional operations, we had to interview farmers every two weeks. We could visit two farmers in one day and, we needed two days every week to check the data and to fill the database. Moreover, to ensure the best quality of data, three feasibility criteria were added to the typology: (1) selecting farmers able to provide detailed information on their practices, (2) selecting tomato fields at the beginning of the crop cycle and, (3) selecting fields with healthy crops to ensure as much as possible to reach the harvest stage. The resulting sample of 12 fields was the only possible sample complying with the need for studying the broader diversity of cropping systems and the feasibility criteria (see Annex 2).

For these 12 cropping systems, we collected data in two steps. First, farmers were asked to describe systematically each field operation: date, plot concerned, container, measure and machine used. To guarantee the accuracy of the data (no language or cultural misunderstanding issues) farmers' declarations were collected by advisors from the communal extensions services involved in the

partnership. Then, a second step consisted in quantifying input flows. Another operator was in charge of expressing the farmer's declarations into quantified flows expressed in standard units (using timer, scales and measuring cups) per hectare (measuring the area of concerned plots).

2.2.3. Cropping systems data for environmental inventories

Inventory data were presented for each modality of each criterion represented in our sample and for their weighted mean. Data collected from the 12 cropping systems in Benin were aggregated into modalities with horizontal averaging, i.e. each parameter was considered independent. To obtain the most representative scenario for the whole population we first averaged data from cropping systems belonging to the same type obtaining 6 virtual scenarios. Then, the weighted mean was calculated as the weighted average of these averages for the 6 virtual scenarios according to the relative weight of their respective corresponding type at population scale on an area-basis. For each virtual scenario, its relative weight was calculated as the ratio between the cultivated area of the specific type and the sum of the cultivated areas of corresponding types within the population. To analyze the dispersion of cropping systems around this weighted mean we calculated the coefficient of variation (CV).

As recommended by Audsley et al. (1997) for an LCA of agriculture, nutrient balances were calculated using the steady state hypothesis of organic matter stability (nitrogen from soil organic matter was neglected). Basically, an element balance equation for any system can be stated as (Oenema et al., 2003): $\Delta P_E = I_E - O_E$ **Eq. 1**

where ΔP_E, I_E and O_E stand for the change in the pool, the input and the output of element E, respectively. Applied to our systems for nitrogen and phosphorus, Eq. (1) can be modified to Eq. (2) with inputs at the left hand and output at the right hand:

$$MinFert_{N\&P} + OrgFert_{N\&P} + Rain_{N\&P} + Irr_{N\&P} =$$
$$Export_{N\&P} + Volat_{NH3} + Denit_{N2O} + Leach_{NO3\&P} + Denit_{N2} + \Delta Soil_{N\&P} \quad \textbf{Eq. 2}$$

Nitrogen and phosphorus inputs from mineral ($MinFert_{N\&P}$) and organic fertilizers ($OrgFert_{N\&P}$) were calculated based on farmer's declaration referring to quantity of fertilizers applied and the nutrient content of these fertilizers. For our systems we set nutrient inputs from rain ($Rain_{N\&P}$) and irrigation water ($Irr_{N\&P}$) to 0. This amount could be significant if waste water was used for irrigation (Diogo et al., 2010). Fresh yields were recorded according to farmer's declaration. Due to the lack of measured data, the exportation of nutrients in harvested tomatoes ($Export_{N\&P}$) were assumed to content 0.15% N and 0.05% P of fresh matter (Diogo et al., 2010). Fertilizer emissions were also estimated, using the best available methods used in LCA. For ammonia emissions from volatilization of mineral fertilizers and poultry manure ($Volat_{NH3}$), emission factors from Bouwman and Van Der Hoek (1997) were used

since they correspond to tropical conditions (4% for NPK, 20% for poultry manure and 25% for urea). We used emission factors from IPCC (2006) to estimate direct (1% of nitrogen inputs) and indirect (1% of NH_3 emitted and 0.75% of NO_3 emitted) nitrous oxide emissions from denitrification (Denit $_{N20}$) and to estimate nitrate leaching ($Leach_{NO3}$ being 30% of nitrogen inputs). Given the lack of specific data on phosphorus emissions in tropical conditions, we followed the recommendations from Nemecek and Kägi (2007) to estimate phosphorus emissions from leaching and run-off (Leach $_{P}$). A part of the nitrogen inputs is emitted as N_2 through denitrification (Denit $_{N2}$) with no potential impact on the environment. To estimate this pool, we applied the emission factor of 9% of nitrogen input as suggested in Brentrup et al.(2000). The remaining nutrient pool in the soil (ΔSoil $_{N\&P}$) was further called nutrient surplus.

For pesticides, applied commercial products were composed by either a unique active substance or by a mix of 2 active substances (ex. dimethoate / cypermethrin). Amounts of products applied per crop cycle were recorded according to farmer's declaration. We calculated the treatment frequency index (TFI) per crop cycle for each pesticide corresponding to the sum of official European approved rates applied. In practice, we divided the amount of active substance applied by its official rate. This indicator allowed us to compare pesticide uses regardless of the nature of the product or the number of actual applications. For each product, a TFI above 1 either meant that more than one application was done or that the product was applied at an excessive rate. Generally speaking, the total TFI per target pest informs about the intensiveness of practices, in terms of number of equivalent official rates. Values for official European approved rates (OR) of each pesticide are provided to allow potential LCA practitioners to calculate the pesticide inputs per hectare.

2.2.4. Comparison with scenarios based on the literature

The comparison and the validation of inventory data are essential for producing reliable LCA(Basset-Mens et al., 2007). Inventory data for our six cropping system types and their weighted mean were compared with agronomic data and measured field emissions from the literature. Seven published articles were identified as suitable for comparison, from which we built three scenarios.

We called the first one "Eritrea scenario", based on agronomic data collected by Asgedom et al. (2011) from a survey of 146 tomato producers located in five regions in Eritrea. Those five regional tomato systems differed from our case study as they were not exclusively located in urban gardens and some were partly rain fed. However, it is to our knowledge the only dataset providing crop lengths (nursery + plot) and crop densities associated to yields. We aggregated (simple average) data related to these five tomato systems under the "Eritrea scenario".

The second comparative scenario was the "Tomato scenario" based on two studies (Diogo et al., 2010; Lompo, 2012). Diogo et al. (2010) collected agronomic data from a 24-month monitoring of ten urban gardens at Niamey (Niger) between 2006 and 2009. Lompo (2012) collected agronomic data from a 24-month monitoring of two urban gardens (Kodéni and Kuinima) at Bobo Dioulasso (Burkina Faso) between 2008 and 2010. These systems were very similar to our case study, even though tomato was preferably grown during the rainy season (March – July) and was irrigated using cans. We extracted from these two studies data related to six tomato crop cycles (Niger: three high input cycles and one low input cycle; Burkina Faso: two cycles from Kodéni) and aggregated them under the "Tomato scenario".

The last scenario was the "Rotation scenario", based on three studies dedicated to emission assessments (Predotova et al., 2010; Predotova et al., 2011; Lompo et al., 2012). These authors measured gaseous or leached emissions for an annual succession of crops from a selection of monitored urban gardens from the previous studies, namely Goudel and Yantala in Niger, and Kodeni and Kuinima in Burkina Faso. For comparison with our data we assumed that a 3-months tomato cropping cycle (as in our Beninese case study) corresponded to 25% of the length of an annual rotation (4 crop cycles on average). We extracted and aggregated data related to these four urban gardens and divided them by a factor 4 under the "Rotation scenario".

2.2.5. Assessment of the quality of LCI data

We applied to our dataset for urban tomato from Benin the LCA-dedicated procedure recommended by most recent guidelines to assess the quality of the LCI data (ILCD handbook, European Commission, JRC, 2010). This procedure is based on the so-called pedigree matrix including five quality criteria proposed by Weidema and Wesnæs (1996): Reliability, Completeness, Temporal correlation, Geographical correlation and Technological correlation.

2.3. Results and discussion

2.3.1. Diversity of cropping systems sampled

Using the three criteria of the typology, 10 cropping system types representative for the diversity of out-of-season tomato cropping systems were identified over the 40 farmers of the population (Figure 2.2). The distribution of cropping system types among the population was uneven in terms of cultivated area. As a result, most modalities were represented by several cropping systems sampled allowing the calculation of within-group variations except for the Sprinkler modality, which was excluded from the aggregation procedure. The 12 cropping systems were sampled within six cropping system types: three belonged to T1, one to T3, one to T5, four to T6, two to T8 and one to T10 (Figure 2.2). Virtual scenarios were calculated for those 6 cropping system types (S1, S3, S5, S6, S8 and S10) and were aggregated into a weighted mean according to their weight at population scale (Figure 2.2).

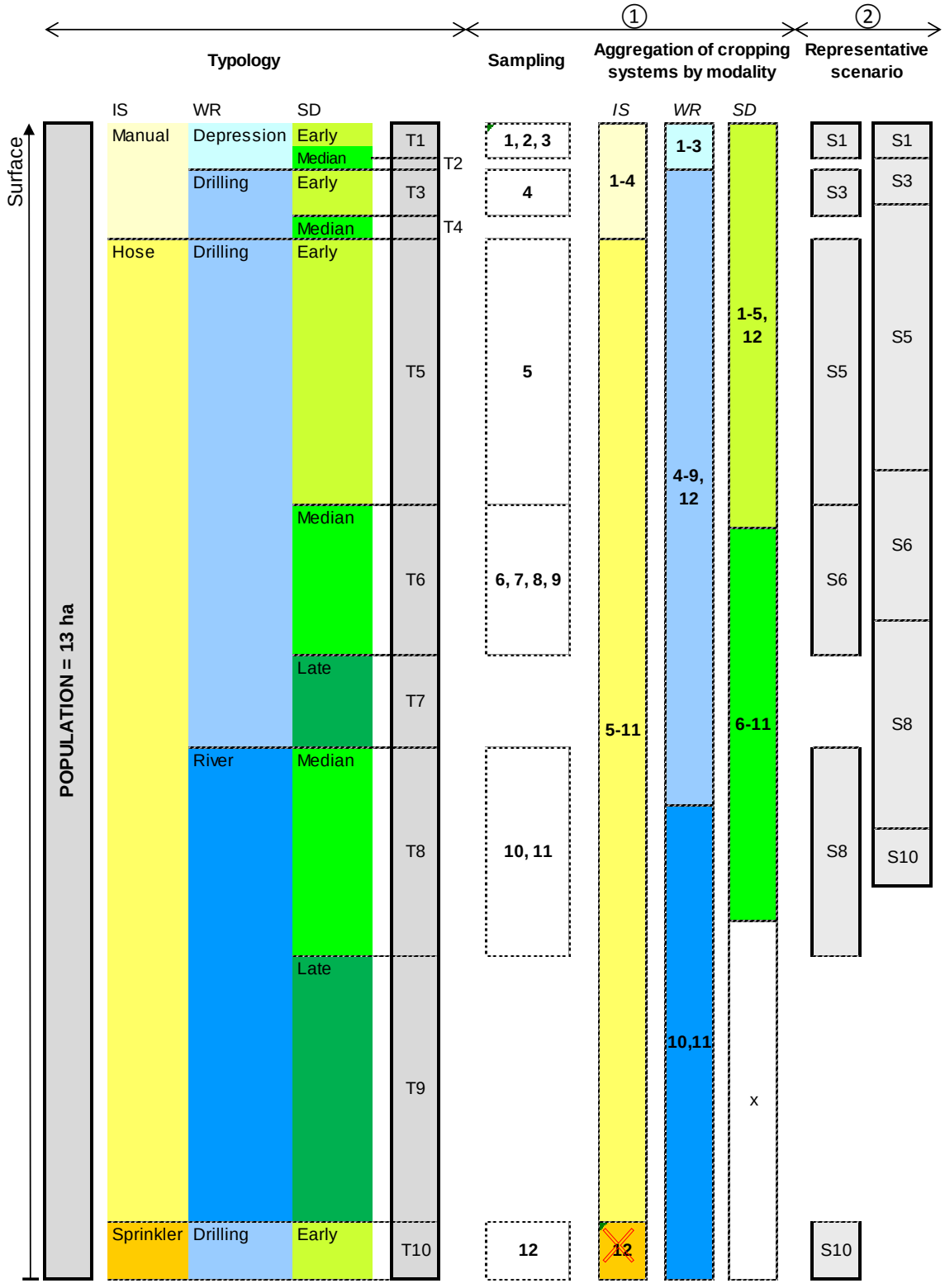


Figure 2.2 The typology-based procedure to (1) sample 12 diverse cropping systems and (2) create a representative inventory for out-of-season tomato cropping systems in studied urban areas in Benin (2011-2012 - Cotonou, Ouidah, Grand Popo). Our typology was based on three criteria easily observable and quantifiable on farms: irrigation system (IS), water resource (WR), and planting date (PD) of the tomato plots. We ended up with 10 cropping system types (T1 to T10). Twelve cropping systems (1 to 12) were sampled within 6 types, for further analysis. Based on sampled types, 6 virtual scenarios were built (S1, S3, S5, S6, S8 and S10) and used to calculate the representative scenario (weighted mean on an area-basis).

Four types were excluded from the sample. T2 and T4, with manual irrigation and crop cycles occurring during the median period, had high pest attacks at the nursery stage and crops were not transplanted. Moreover, late crop cycle types (T7 and T9) were not advanced enough at the time of the data collection process and were therefore discarded. Regarding geographical diversity, T1, T3, T5 and T10 were mainly represented by farmers at Cotonou or Pahou while T6 and T8 were mainly represented by farmers at Grand-Popo (Table 2.1). At sample scale we observed a gradient of farm size across cities, resulting from high land pressure at Cotonou to low land pressure at Grand-Popo. To further investigate the effect of land pressure, being one of the characteristics of urban agriculture, we will add this criterion for cropping system aggregation and inventory analysis.

Table 2.1 Distribution of the 10 types at the whole population scale (typology based on the irrigation system, the water resource and the planting date) in terms of area (m²) and number of farms for each district

Irrigation system	Water resource	Planting date	Cotonou		Ouidah		Grand Popo		Population		Type
			Area (m²)	Farms	Area (m²)	Farms	Area (m²)	Farms	Area (m²)	Farms	
Manual	**Depression**	**Early**	1278	5	320	1			4478	6	1
		Median	702	1					702	1	2
	Drilling	**Early**	1650	2	400 0	2			5650	4	3
		Median	1575	4	800	1			2375	5	4
Hose	**Drilling**	**Early**			288	7	1800	3	30600	10	5
		Median	90	1	161	3	400	1	16610	5	6
		Late					11144	3	11144	3	7
	River	**Median**			440	1	20000	1	24400	2	8
		Late					30000	1	30000	1	9
Sprinkler	**Drilling**	**Median**	2064	2			4000	1	6064	3	10
Total			7359	15	573	15	67344	10	13202	40	

We present in sections 2.3.2. to 2.3.4 our field data aggregated by modality for each criterion and the weighted mean compared to the data of the "Eritrean scenario", the "Tomato scenario" and the "Rotation scenario" based on the literature (Diogo et al., 2010; Predotova et al., 2010; Asgedom et al., 2011; Predotova et al., 2011; Lompo, 2012; Lompo et al., 2012). The specificity of our studied tomato cropping systems was to be out-of-season, i.e. occurring during the dry season, while tomato cropping systems from the literature scenarios occurred mostly during the rainy season. Our data were representative for only one crop cycle of tomato, while the "Rotation scenario" was representative for one climatic year during which several vegetable cropping systems occurred.

2.3.2. Crop protection: frequent applications of pesticides at excessive rates

Crop protection practices revealed high and variable treatment frequency (TFI) indexes especially for insect control. High TFI values highlighted both high frequencies of application (Table 2.2) and the application of excessive rates compared to official approved rates. Farmers systematically applied insecticides (CV=73%accross cropping systems, Table 2.2) while the use of pesticides for disease and nematode control was more occasional (CV=78% and 127% respectively).

Table 2.2 Average number of applications and associated coefficient of variation per targeted pest: insect, disease and nematod for cropping systems aggregated by modality for each criterion: irrigation system, water resource and planting date and for the scenario representative for the whole population (WM).

Criterion	Modality	Insect		Disease		Nematod	
		Average	*CV*	*Average*	*CV*	*Average*	*CV*
Irrigation system	**Manual**	5.3	75%	2.5	124%		-
	Hose	2.4	62%	1.4	114%	0.4	124%
Water resource	**Depression**	3.7	79%	3.3	96%		-
	Drilling	3.2	103%	1.4	114%	0.3	170%
	River	4.0	0%	1.5	141%	1.0	0%
Planting date	**Early**	4.5	72%	2.8	87%	0.2	245%
	Median	2.3	70%	1.0	126%	0.5	109%
Location	**Cotonou**	5.5	67%	2.8	113%	0.3	200%
	Grand-Popo	2.3	89%	2.4	89%	0.3	173%
	Ouidah	2.4	47%	1.0	100%	0.4	137%
Representative scenario	**WM**	3.7	73%	2.7	78%	0.4	127%

For insects' control (Figure 2.3), the weighted mean presented a treatment frequency index of 8.2 (CV= 85%). Cropping systems with manual irrigation showed higher TFI than cropping systems with hose irrigation (12.0 vs. 5.7) systems. Similarly, early cropping systems showed higher TFI than median ones (9.9 vs. 5.1) and cropping systems from Cotonou showed higher TFI than that from other locations (11.6 vs. 4.9 and 6.3 respectively). The highest TFI (manual irrigation systems) resulted from frequent applications at excessive rates per application (5.3 applications at 2.3 fold the official rate on average, Table 2.2). High TFI resulted mainly from the use of pyrethroids, i.e. cypermethrin, lambda cyalothrin and deltamethrin (red colors vs. purple colors).

For diseases control (Figure 2.3), the weighted mean presented a TFI of 1.8 (CV=92%). Similarly to insect control, the modalities manual irrigation systems, water from depression and Cotonou showed higher TFI than other modalities within the same criterion. The highest TFI (manual irrigation with TFI = 3.4) resulted more from frequent applications than excessive rates per application (3.4 applications at 1.4 fold the official rate on average, Table 2.2).High TFI resulted from several uses of maneb, thiophanate methyl or mancozeb (blue colors vs. green colors).

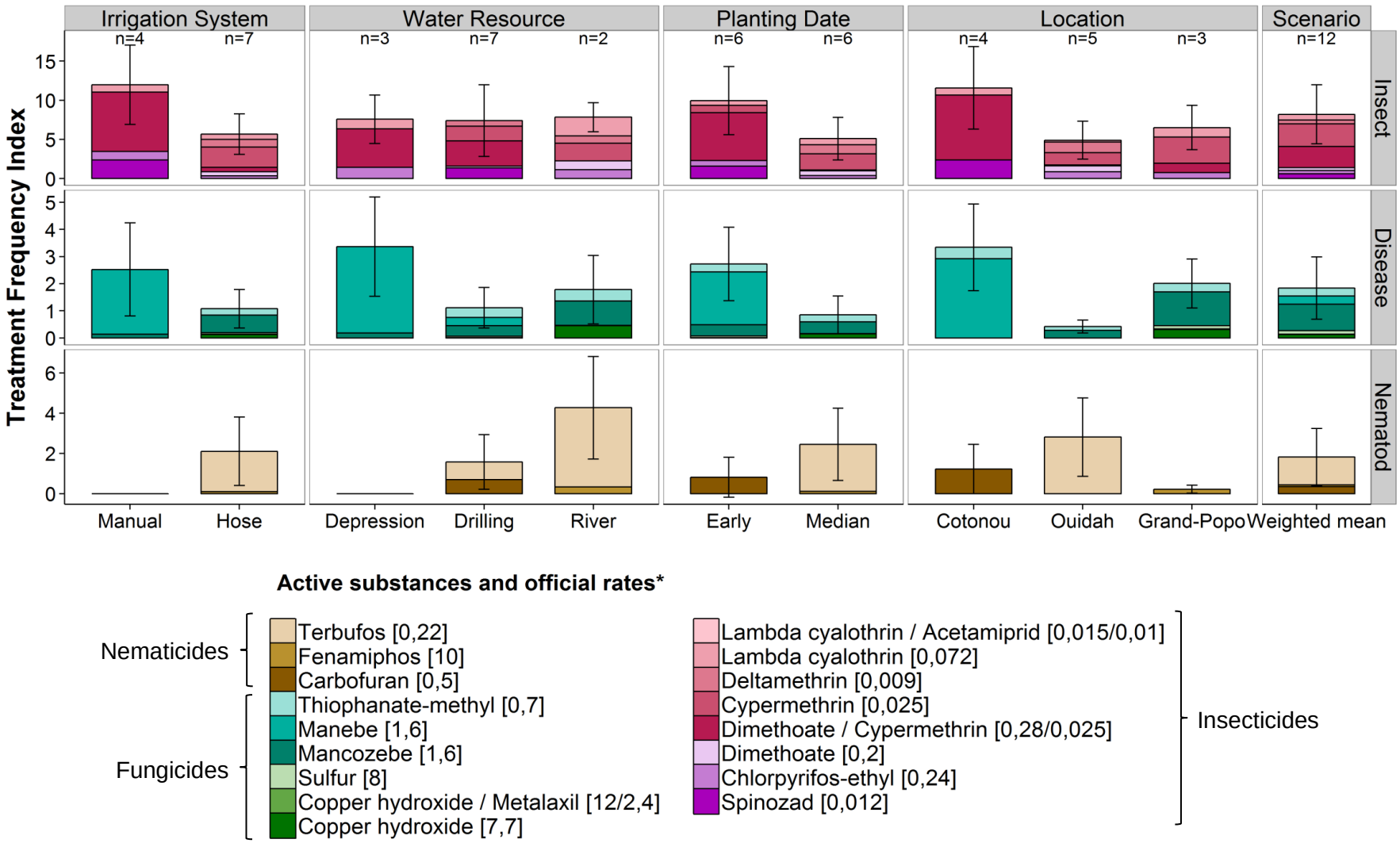


Figure 2.3 Treatment frequency index (TFI) per targeted pest: insect, disease and nematod for each active substance used for crop protection, for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean). *Official European approved rates expressed in kg of active substance per hectare were used for calculation of TFIs.

For nematodes control (Figure 2.3), the weighted mean presented a lower TFI (TFI =1.8) compared to insect control but was highly variable across cropping systems (CV= 156%) partly due to the absence of treatments for cropping systems with manual irrigation. Cropping systems using water from river showed higher TFI than cropping systems with drilling (4.3 vs. 1.6). Similarly, median cropping systems showed higher TFI than early ones (2.5 vs. 0.8). The highest TFI (water from river) resulted from an excessive rate per application (1 application at 4.3 fold the official rate, Table 2.2). The highest TFI were associated to the use of carbofuran and terbufos. It seems that the choice of the active substance to control nematodes depends on the location.

The orders of magnitude of treatment frequency index found in our study are in agreement with the high number of pesticide applications recorded from studies based on a larger sample of urban gardeners (superior to 100) in South-Benin (Ahouangninou et al., 2011) and Senegal (Cissé et al., 2003). To our knowledge this is the first detailed dataset on crop protection practices for tomato cropping systems in Africa. As suggested by (Ahouangninou et al., 2011) and (Cissé et al., 2003), pest control practices were likely to result from a lack of knowledge and technical support. Farmers struggle to identify the pests and to master instructions for application. We noticed that high TFI were often associated to active substance with low official rates (pyrethroids, thiophanate methyl, maneb, mancozeb, terbufos and carbofuran). It suggests that farmers had difficulties to apply pesticides at low rate, most probably due to inaccurate measuring tools and small plots. This finding was in agreement with observations made on the excessive use of Endosulfan (insecticide) for a large sample of urban gardeners in South-Benin(Ahouangninou et al., 2012). According to Ahouangninou et al.(2012), the chlorpyrifos ethyl, the pyrethroids and the mancozeb present particularly high toxicological risks (as a result of high environment and health risk index). Owing to the wide use of pesticides on out-of-season tomato in Benin, our inventory study should be completed with a proper environmental assessment including impacts on freshwater and terrestrial ecotoxicity.

2.3.3. Intensive fertilization management: important N and P surplus

N and P inputs

Overall, N and P inputs as organic and mineral fertilizers were large and in the same range as that for the literature scenarios (Figure 2.4). More specifically, the weighted mean presented similar total nitrogen inputs to the "Tomato scenario" (309 kg $N.ha^{-1}$) and a lower total nitrogen inputs than the "Rotation scenario" (449 kg $N.ha^{-1}$), due to lower amounts of mineral nitrogen. Unlike for the nitrogen, the weighted mean presented a higher amount of total phosphorus inputs than the "Tomato scenario" (104 vs. 68 kg $P.ha^{-1}$), due to higher amounts of organic phosphorus. The total phosphorus input for the weighted mean was slightly lower than the total P input for the "Rotation scenario" (102 kg $P.ha^{-1}$). It seems that the relatively high phosphorus content of poultry manure was responsible for high phosphorus inputs in our Beninese systems.

The analysis of nutrient input variations between modalities within each criterion was hampered by large within-modalities variations (up to for 101% for the Depression modality). However, it appeared that cropping systems from Cotonou showed higher amounts of nitrogen and phosphorus inputs compared to other locations (711 vs. 447 and 122 kg $N.ha^{-1}$respectively). This finding pleads for a correlation between the location of cropping systems, as an indicator of land pressure, and the level of intensification for fertilization practices. The contribution of mineral fertilizer to the total nitrogen input varied from 3% (Grand-Popo) to 86% (Ouidah) and from 8% (Grand-Popo) to 83% (water from river) for the total phosphorus input. For mineral fertilization, the three same types of fertilizer were recorded across our types and across literature scenarios: NPK, NP and urea. The uneven contribution pattern of the different nitrogen origins across cropping system types, suggests that the access to fertilizers was most likely another factor driving management practices.

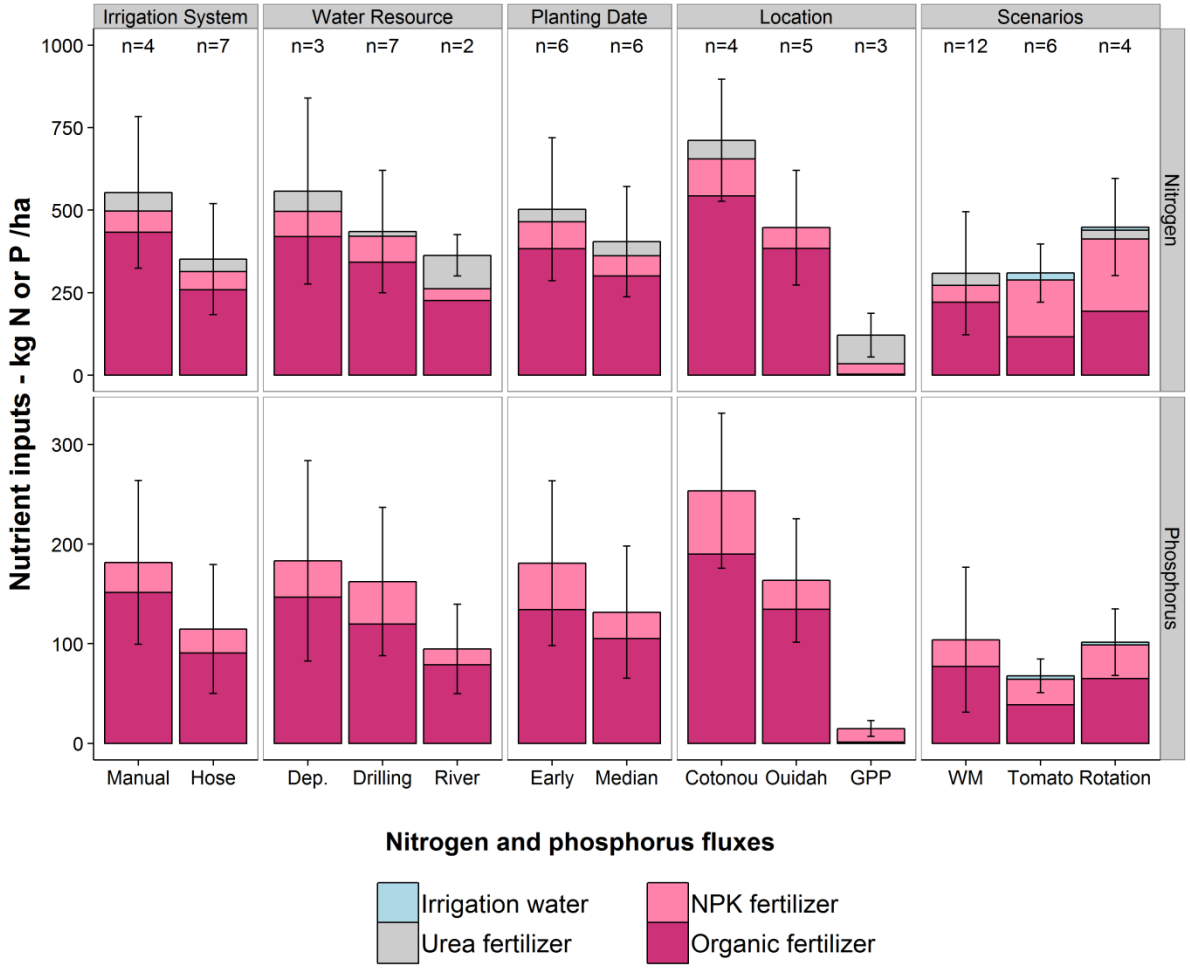


Figure 2.4 Nitrogen and phosphorus inputs for main nutrient horizontal fluxes at field scale; for cropping system aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean). We compared them to two scenarios from the literature: the "Tomato scenario" corresponds to the average of 6 tomato cropping systems from urban gardens in West Africa (Diogo et al., 2010; Lompo, 2012) and the "Rotation scenario" corresponds to the average of 4 annual rotations from urban gardens in West Africa (Predotova et al., 2010; Predotova et al., 2011; Lompo et al., 2012).

N and P outputs

Generally speaking, nutrients exportations through harvest were very low compared to inputs but were larger than that for the "Tomato scenario" (Fig 2.5). More specifically, the weighted mean presented slightly higher nitrogen exportation than the "Tomato scenario" (14 vs. 10 kgN.ha^{-1}). However, the weighted mean for nitrogen exported with harvest was lower than that for the "Rotation scenario" (124 kg N.ha^{-1}). Similar pattern was observed for phosphorus exportations. These differences resulted from the lower nutrient content of tomato compared with other vegetables as included in the "Rotation scenario".

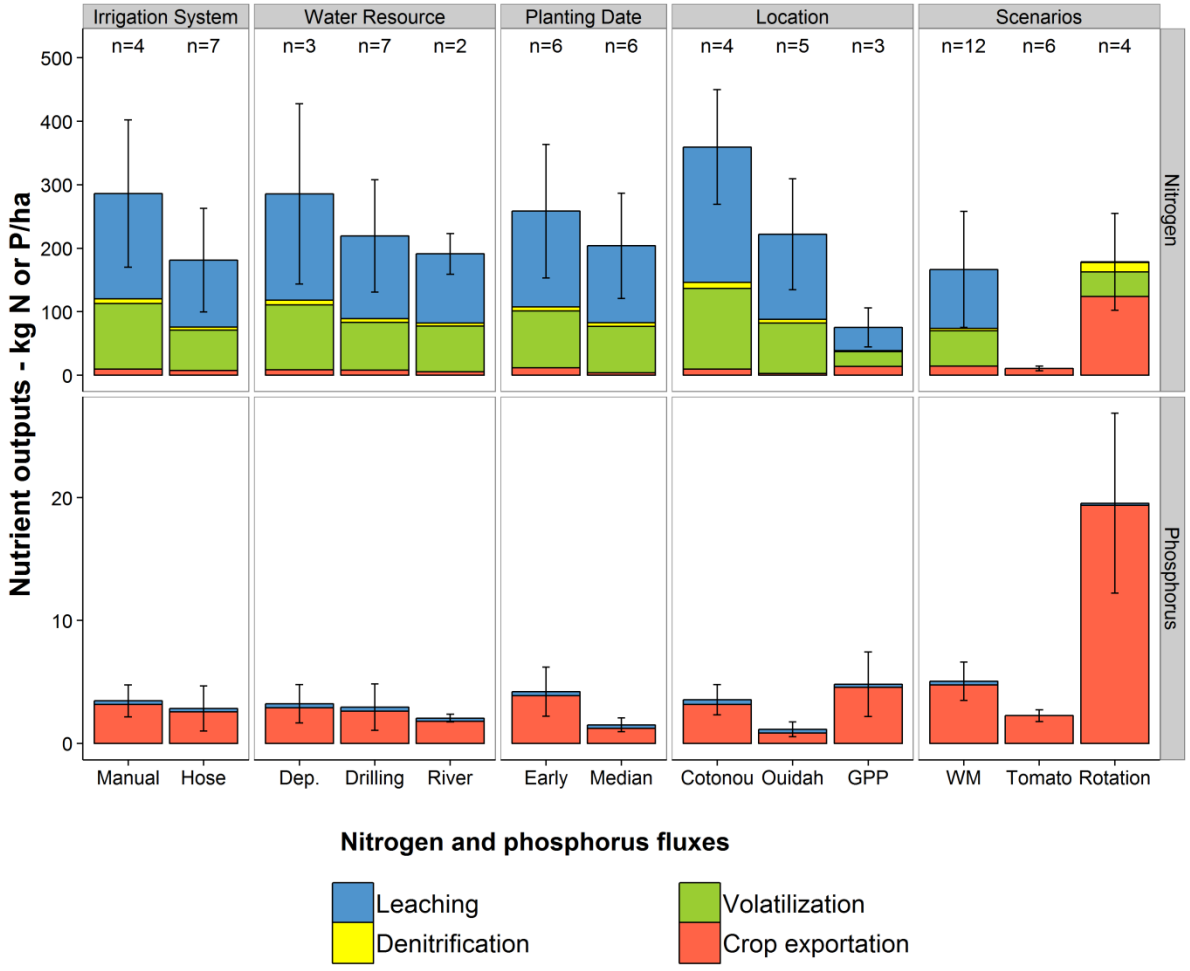


Figure 2.5 Nitrogen and phosphorus outputs for main nutrient fluxes at field scale for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean). We compared them to two scenarios from the literature: the "Tomato scenario" corresponds to the average of 6 tomato cropping systems from urban gardens in West Africa (Diogo et al., 2010; Lompo, 2012) and the "Rotation scenario" corresponds to the average of 4 annual rotations from urban gardens in West Africa (Predotova et al., 2010; Predotova et al., 2011; Lompo et al., 2012).

As a rule, estimated nitrogen emissions were large especially through nitrate leaching and ammonia volatilization (Figure 2.5) compared to the "Rotation scenario". It is worth noticing that emissions were not measured for the "Tomato scenario". More specifically, cropping systems in Benin showed greater nitrogen losses from leaching than from volatilization and denitrification. The weighted mean presented a lower emission factor for NH_3 than the "Rotation scenario" (15 vs. 39% of the total nitrogen inputs, Table 2.3), despite the high emission factors applied to estimate emissions from urea and manure taken from a study dealing with tropical contexts (Bouwman and Van Der Hoek, 1997). However, observed differences between our NH_3 estimates and measured NH_3 emissions from the literature were probably not significant compared to the variability of the phenomenon at local scale (Table 2.3). The weighted mean presented a lower emission factor for N_2O than the "Rotation scenario" (1 vs. 15% of the total nitrogen inputs). By contrast, the weighted mean presented a higher emission factor for NO_3 than the "Rotation scenario" (30 vs. 1% of the total nitrogen inputs). Using estimation methods in our case study most likely led to an underestimation of N_2O emissions and an overestimation of NO_3 leaching even considering variations. Finally, phosphorus emissions were very low for our study as for the "Rotation scenario". However, this does not anticipate the associated potential impact on eutrophication, phosphorus being a limiting factor in natural ecosystems.

Owing to the amount of nitrogen emitted to air and water and potential risks associated to P losses, our inventory study must be completed with a proper environmental assessment including impacts on both acidification and eutrophication potentials.

Table 2.3 Average and range of variation for emission factors for the scenario representative (weighted mean) where emissions were estimated based on the literature and for the Rotation scenario where emissions were measured on field (18-month period)

Emission factors	**Weighted mean**		**Rotation**	
kg N or P.100 kg^{-1} N or P applied	*Average*	*Variations*	*Average*	*Variations*
N-NH_3	15.0%	[4.5-25]	39.0%	[7.2-75.4]
N-N_2O	1.3%	-	14.9%	[5.0-27.1]
N-NO_3	30.0%	-	1.0%	[0.5-1.4]
P-PO_4	0.9%	-	0.7%	[0.2-1.1]

Nutrients surplus

Figure 2.6 showed that despite the inclusion of all emissions, the nitrogen and phosphorus surplus remained large and similar to surplus observed in the literature (Diogo et al., 2010; Lompo, 2012). This surplus resulted from high inputs compared to crop exportations. It suggests that emissions could occur beyond the crop cycle duration in relation to the large pools of N and P stored in soils. The

commonly used assumption of stability of the soil organic matter appears questionable for tomato urban gardens which are often recently settled and intensified cropping systems. This result raises the question of the fate of these nutrients surplus after harvest. It also invalidates the hypothesis of organic matter stability at cropping system scale. Our nitrogen budget also showed that neglecting N_2 emissions as was the case in the reviewed literature, can lead to large underestimations of nitrogen outputs.

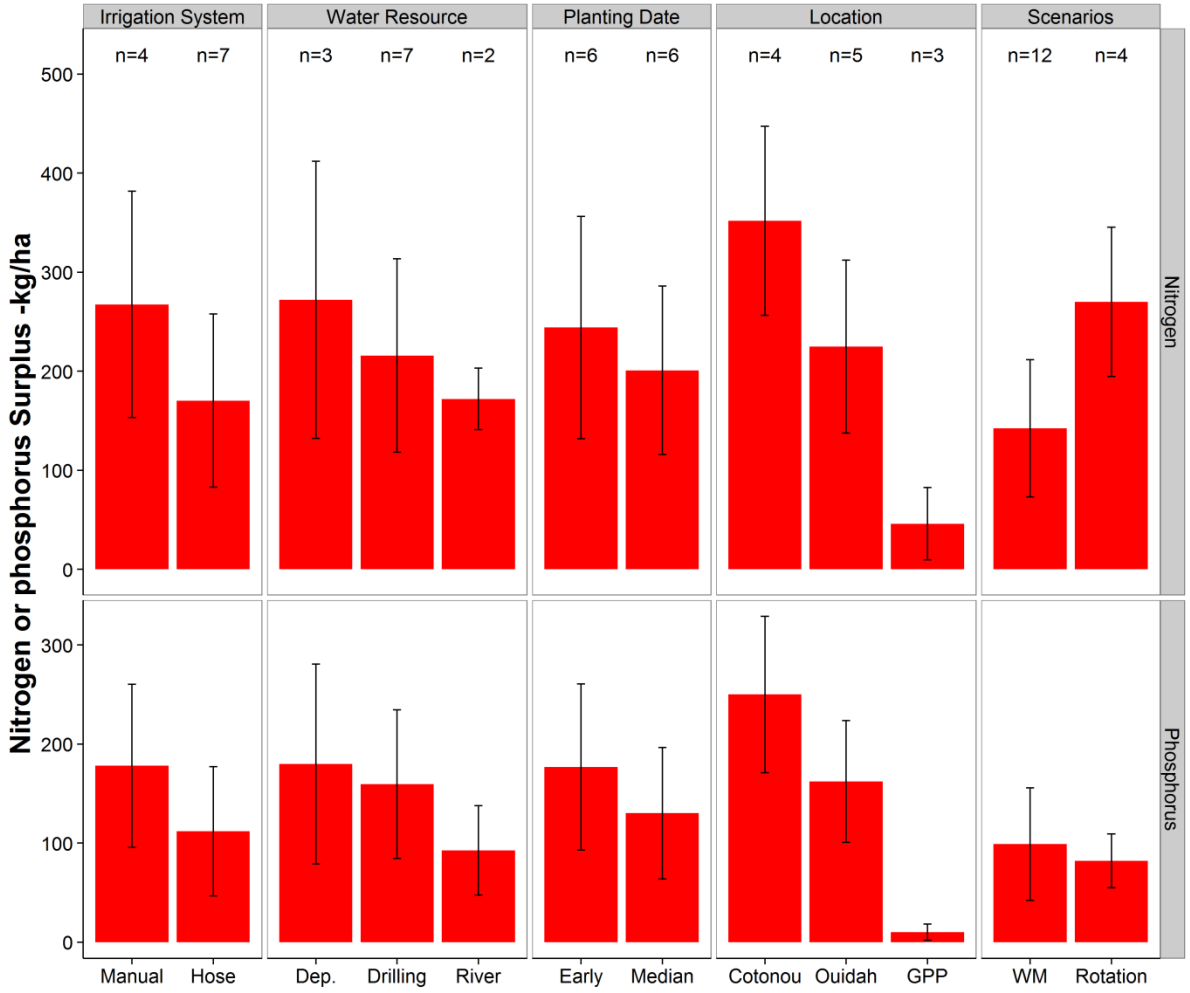


Figure 2.6 Nitrogen and phosphorus surplus at field scale for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (weighted mean). We compared them to two scenarios from the literature: the "Tomato scenario" corresponds to the average of 6 tomato cropping systems from urban gardens in West Africa (Diogo et al., 2010; Lompo, 2012) and the "Rotation scenario" corresponds to the average of 4 annual rotations from urban gardens in West Africa (Predotova et al., 2010; Predotova et al., 2011; Lompo et al., 2012).

2.3.4. Variable and low tomato yields

Overall, the crop yields for Beninese tomato were low and variable but in the same range as the yields observed in literature scenarios (Table 2.4). The weighted mean presented a slightly lower yield than the "Tomato scenario" (9,533 vs. 10,342 $kg.ha^{-1}$) and a slightly greater yield than that the "Eritrean

scenario (7,300 kg.ha^{-1}). However, any modality reached the yield of the weighted mean which results from the high weighting factor associated to a cropping system presenting a yield twice as great as other yields within the sample. The location and the planting date appear to be the most discriminant criteria, with early cropping systems reaching higher yields than median cropping systems and cropping systems from Ouidah showing the lowest yields.

The plot density weighted mean for our types was similar to that for the "Eritrean scenario" (4.9 vs. 5.4 plants.m^{-2}). However, compared to similar tomato cropping systems from the literature (Asgedom et al., 2011), the short crop lengths (78 vs. 105 days) indicated that farmers had to reduce their crop cycles, limiting the potential yield. Cropping systems from Ouidah showed short crop lengths and the lowest yields, due to random pest attacks. If no effective pesticide for these massive attacks was available (*Tetranychus evansi* also called the red spider mites for example) the farmer may lose the whole harvest.

Table 2.4 Cropping system data over one tomato crop cycle for cropping systems aggregated by modality for each criterion: irrigation system, water resource, planting date and location and for the scenario representative for the whole population (WM). We compared them to two scenarios from the literature: the "Eritrea scenario" corresponds to the average of 5 tomato cropping systems in Eritrea (Asgedom et al., 2011), the "Tomato" corresponds to the average of 6 tomato cropping systems from urban gardens in West Africa (Diogo et al., 2010; Lompo, 2012). Some data were not available in reviewed papers (na).

Criterion	**Modality**	**n**	**Nursery length** (days)		**Plot lenght** (days)		**Plot density** (plants.m^{-2})		**Fresh yield** (kg.ha^{-1})	
			Average	*CV*	*Average*	*CV*	*Average*	*CV*	*Average*	*CV*
Irrigation system	**Manual**	4	43	33%	76	10%	3.4	44%	6309	83%
	Hose	7	23	28%	74	28%	4.2	41%	5123	144%
Water resource	**Depression**	3	36	15%	75	12%	3.7	48%	5787	108%
	Drilling	7	29	58%	76	26%	4.1	43%	5219	147%
	River	2	28	15%	68	23%	3.4	2%	3604	35%
Planting date	**Early**	6	38	37%	77	13%	4.1	48%	7733	105%
	Median	6	23	30%	71	29%	3.7	30%	2450	94%
Location	**Cotonou**	4	45	28%	73	11%	3.9	27%	6307	83%
	Ouidah	5	23	33%	69	32%	3.0	28%	1701	147%
	Grand-Popo	3	25	18%	85	8%	5.5	37%	9121	115%
Scenarios	**WM**	12	27	48%	78	20%	4.9	32%	9533	66%
	Eritrea	5	31	13%	105	14%	5.4	30%	10342	26%
	Tomato	4	na	na	na	na	na	na	7300	64%

2.3.5. Prospects for the Life cycle assessment of out-of season tomato in Benin

Using the five quality criteria of the pedigree matrix from Weidema and Wesnæs (1996), we obtained good scores for our data set (Table 2.5).

Reliability: The raw data were collected by local advisors directly from farmers' declarations. Most farmers could provide detailed technical data although they had no records. Due to the diversity of measure units used by farmers, the difficulty relied in the conversion of farmer's declarations into quantified data expressed in standard units per hectare to obtain a homogeneous dataset. The quantification of input flows (section 2.2.2) on the farm allowed a satisfactory level of data reliability regarding the pedigree matrix criteria.

Completeness: We collected a complete dataset describing agricultural practices and resource use from sowing to the last harvest. The cropping system selection allowed us to reach a fair representativeness of the practices from the 40 identified farmers while maintaining a reasonable number of cropping systems monitored with regard to time and labor constraints. With a capacity of monitoring 12 cropping systems, we could only cover 30% of existing cropping systems (on the basis of 40 farmers = 40 cropping systems). However, 70% of the farmers presented one of the six cropping system types studied (Table 2.1) which represented 60% of the cultivated area of our farmers' sample. A first limitation of our dataset is the sample size compared to the high diversity of cropping systems within the population. Some types being represented by only one cropping system, it excluded the possibility to analyze the cropping systems' variability within each type. We thus analyzed the effect of modalities for each criterion independently. We highlighted interesting trends regarding practices across modality and the weighted mean succeeded in providing an average profile regarding most of the criteria. However, it appeared that the representative scenario (weighted mean) led to an overestimation of the average yield for the population (Table 2.4). The second limitation of our dataset refers to the lack of consideration for annual variations due to climate or socio-economic parameters. However, compared to the scenarios based on the literature aggregating systems from several years and gardens, the weighted mean showed comparable and often higher coefficients of variation. We therefore most likely succeeded in capturing an important part of the inherent variability of cropping systems through our typology.

Temporal, geographical, and technological correlations: The studied cropping systems presented the three technical modalities of irrigation systems and sources of water. The distribution of modalities within the population was respected in the sample, with a small over-representation of hose systems and depression-sourced water systems. The main limitations were first our inability to study cropping systems representative for the late planting date modality and second, the reduced number of cropping systems with sprinkler irrigation system.

Following our typology, we succeeded at producing good quality data on cropping systems producing out-of-season tomato in Benin to be used for life-cycle assessment (LCA). Such dataset could only be achieved thanks to strong and multiple partnerships with locally established research organizations, extension services, and farmers. The two first partners provided expertise to cope with the lack of bibliography and selected criteria for the typology. They also provided a precious repertory of farmers which guaranteed an unbiased selection of cropping systems, along with expertise and references to ensure the quality of data. Finally, the farmers were able to answer to our questions although they had no written records.

Table 2.5 Data quality indicators (Weidema and Wesnaes, 1996) for our LCI data set for out-of-season tomato in Benin. The best score is 1, the worst is 5.

Quality Indicator	Score	Data quality goal	Performances and limitations
Reliability	1/5	"Verified data based on measurements"	Farmer's declaration collected by local advisors Straightaway quantification and checking
Completeness	3/5	"Representative data from an adequate number of sites but from shorter periods"	Representative for 70% of farmers' practices The variability between cropping systems types and between year could not be quantified
Temporal correlation	1/5	"Less than three years of difference to year of study"	Data collected especially for the study Homogenous period (out-of-season)
Geographical correlation	1/5	"Data from the area under study"	Representative for 60% of the production area
Technological correlation	3/5	"Data from processes and materials under study but from different technology"	Representative for 60% of types based on technological criteria identified with an expert panel Sprinkler irrigation systems and late crop cycles could not be analyzed

2.4. Conclusions

In this paper, we presented cropping system data for representative environmental inventories of one hectare of out-of season tomato in Benin. We used a typology to sample 12 cropping systems within a population of 40 farmers. We analyzed the different inventory data for the range of modalities for each criterion of the typology and for their weighted mean and compared them to comparable data from scenarios based on the literature. We then assessed the resulting inventory data according to consensual quality criteria as recommended by the ILCD Handbook from the European Commission.

Pesticide and fertilizer rates were variable and excessive. The weighted mean TFI reached 8.2 for insecticides and 1.8 for both fungicides and nematicides. TFI were highly variable between cropping systems and within modalities. However, it appeared that cropping systems with either manual irrigation, early crop cycle or being located at Cotonou presented more intensive pest and disease control practices. Our results confirmed that pest control practices were likely to result from a lack of

knowledge and technical support, as most of high TFI resulted from excessive application rate and/or frequent application of the same substance. It appeared that farmers encountered difficulties at applying pesticides at low rate. Beyond the large field emissions to the environment, the weighted mean nutrient budget showed large nutrient surplus: 120 kg N and 84 kg P. ha^{-1}. These large surpluses resulted from high nutrient inputs compared to crop exportations. Our results suggested a correlation between the location of cropping systems, as an indicator of land pressure, and the level of intensification for fertilization practices. Yields were low (11,501 $t.ha^{-1}$ for the weighted mean) and also variable, due to random pest attacks. Overall, practices were not optimized and revealed low performance and high risks for the environment.

Our inventory data for out-of-season tomato in Benin were of good quality according to commonly used criteria in LCA. Allowing the selection of 12 representative cropping systems over a population of 40 farmers and their aggregation into a weighted mean, the typology allowed covering most of the geographical and technical diversity characterizing the population. The environmental assessment of those production systems should at least include freshwater and terrestrial ecotoxicity, eutrophication, acidification, global warming and water deprivation potentials.

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Contribution du chapitre 2 et perspectives dans le cadre de la thèse

Ce chapitre présente le protocole de collecte des données qui seront utilisées dans la suite de la thèse pour réaliser l'ACV de la tomate produite en jardin urbain au Bénin (Figure 2.7). Pour chaque parcelle suivie, nous avons enregistré pour chaque opération culturale : la date, les produits utilisés et les doses déclarés sous la forme des instruments de mesure utilisés par le producteur: une capsule, une poignée, un sceau (les protocoles d'enquête sont visibles en annexe 3). L'ensemble des données collectées est résumé en annexe 4 et un itinéraire technique complet est proposé en annexe 5. Enfin nous avons également réuni des données pédoclimatiques: 1 analyse de sol par parcelle (Annexe 6) et les données pour la station météo de Cotonou (Annexe 7). Ces données seront reprises pour simuler les rendements et les émissions dans le chapitre 3, puis dans le chapitre 4 (Figure 2.7).

Le chapitre 2 met par ailleurs en application les recommandations du chapitre 1 pour l'estimation des émissions d'azote réactif et aboutit à la définition d'une méthode de facteurs d'émission par défaut (DEF) qui sera ensuite utilisée dans le cadre du chapitre 4 pour produire les résultats d'ACV dit 'de référence' (Figure 2.7). Nous avons également mis en évidence un corpus bibliographique portant sur les bilans de flux de nutriments à l'échelle de la parcelle. Ces références seront reprises dans le chapitre 3 pour analyser la pertinence des émissions simulées par le modèle biophysique (Figure 2.7).

Enfin, ce chapitre permet d'aboutir à un niveau d'agrégation des parcelles basé sur 4 critères discriminants (le système d'irrigation, l'origine de l'eau, la date de repiquage et la localisation géographique) et à un scénario représentatif de la population basée sur la moyenne pondérée. Ces scénarios seront utilisés dans le cadre du chapitre 4 pour produire les résultats d'ACV.

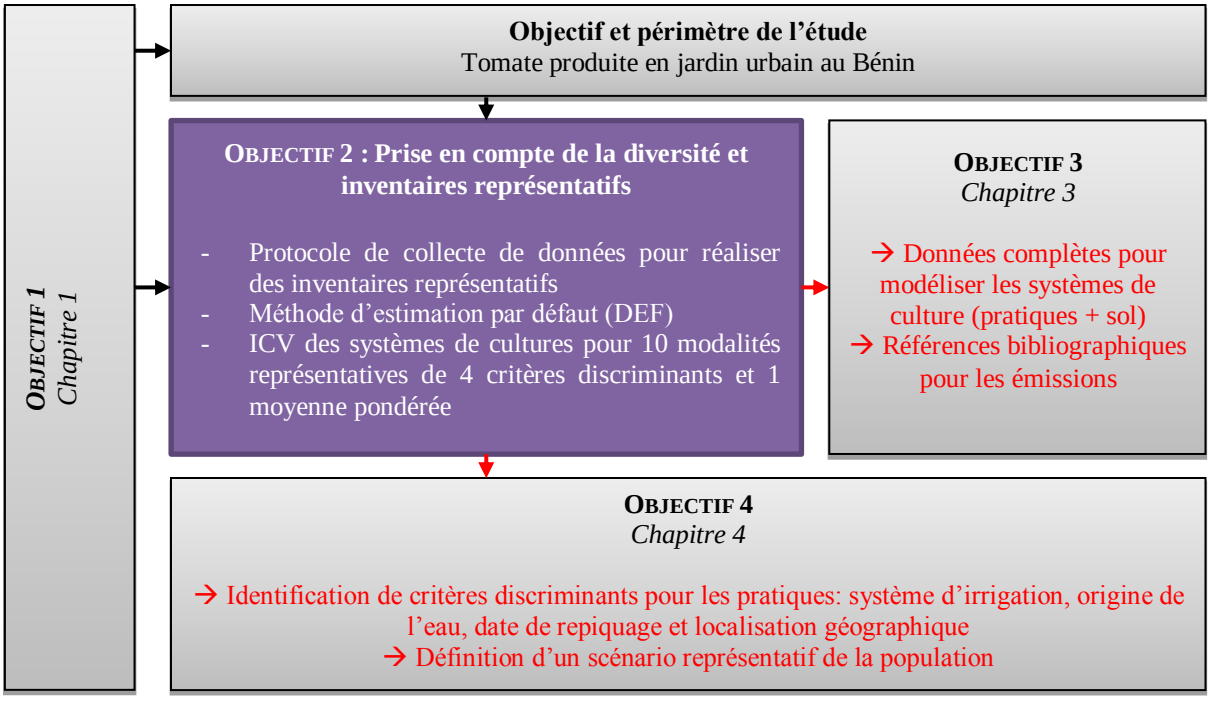


Figure 2.7 Résultats du chapitre 2 (boite violette) et contribution pour les chapitres suivants (rouge)

Chapitre 3. Boucler le bilan d'azote à l'échelle du système de culture : perspectives pour l'estimation des émissions d'azote réactif pour la tomate en maraîchage urbain au Bénin



Carré de récolte sur une parcelle de tomate en jardin urbain à Cotonou, Bénin

Avant-Propos

L'analyse de la bibliographie dédiée aux flux d'azote réactif réalisée dans le chapitre 1 nous a permis d'identifier l'espace de complexité qui existe entre les caractéristiques des systèmes de culture et les émissions. Le tableau 3.1 présente la liste des facteurs susceptibles d'influencer les émissions d'azote réactif dans le contexte d'étude, établie à partir de la bibliographie sur les émissions. Confronter cette liste aux outils de modélisation disponibles, nous a permis d'identifier un modèle candidat permettant d'intégrer un maximum de facteurs influents et pouvant être utilisé avec les données disponibles. Pour notre application, nous avons choisi le modèle STICS car il permettait de simuler l'ensemble des émissions d'azote réactif en plus des exportations d'azote par la plante, en prenant en compte de manière robuste une grande diversité de conditions pédoclimatiques et de pratiques y compris celles de nos systèmes. Nous présentons dans ce chapitre l'exploration de l'usage de ce modèle pour l'ACV de notre cas d'étude.

Tableau 3.1 Facteurs influençant la variation des émissions azotées au champ pour les cultures maraichères

	Facteurs influents	**NH_3**	**N_2O**	**NO_3**
Sol	Taux d'infiltration	✓		✓
	pH, CEC_a	✓	✓	✓
	Teneur en calcaire et en argile	✓		
	Température du sol	✓	✓	
	Taux de matière organique			✓
	$N_{org\ b}$, $[NH_4]_c$, $[NO_3]_c$		✓	✓
	Potentiel de dénitrification	✓		
	Capacité au champ, Point de flétrissement permanant	✓	✓	
	Densité apparente	✓		
Climat	Déficit de saturation /Pression de vapeur saturante	✓		✓
	ETP_d	✓		
	Température de l'air	✓		
	Albédo	✓		✓
Pratiques	$[NH_4]$, $[NO_3]$ des engrais minéraux	✓	✓	✓
	MO, N_{org}, $[NH_4]$, $[NO_3]$ des engrais organiques	✓	✓	✓
	Date et fractionnement de la fertilisation	✓	✓	✓
	Méthode d'incorporation engrais	✓		
	Date et fractionnement de la fertilisation l'irrigation	✓	✓	✓
	Drainage et ruissellement induit par les pratiques			✓
	LAI_e	✓		✓
	N_{org}, $[NO_3]$ des résidus			✓
	Azote absorbé par la plante		✓	✓

a. Capacité d'échange cationique, b. azote organique, c. teneur en azote ammoniacal et nitrique à l'état initial, d. évapotranspiration potentielle, e. Indice foliaire (Leaf Area Index)

Résumé

Dans le but d'évaluer l'impact environnemental des productions maraichères périurbaines en contexte africain, il est nécessaire de connaitre les flux d'azote réactif et leurs interactions avec le climat, le sol et les pratiques. Pour des systèmes sur lesquels les données sont rares et les mesures difficiles à mettre en place, l'approche par modélisation est souvent incontournable. Une approche combinant un bilan d'azote à l'échelle de la parcelle et l'estimation des flux d'azote par un modèle biophysique a été appliquée à nos systèmes de production de tomate au Bénin. Les objectifs de cette approche sont (1) d'identifier les facteurs explicatifs des flux d'azote (2) de comparer les bilans azotés de 12 systèmes de production de tomate (3) d'identifier les incertitudes et de fournir des recommandations pour l'estimation des flux d'azote réactif en contexte tropical.,

La modélisation des flux d'azote réactif a été réalisée à l'aide du modèle STICS, qui a l'avantage d'être utilisable en contexte tropical et de fournir des données de flux détaillées réutilisables pour calculer un bilan d'azote à l'échelle de la parcelle. Les variations entre les entrées et les sorties du modèle ont été analysées à l'aide de régressions linéaires simples et multivariées. Les résultats obtenus montrent la difficulté à modéliser les rendements, qui sont globalement surestimés par rapports aux rendements observés. Ceci s'explique par une forte pression biotique (maladies et ravageurs) mal contrôlée entrainant une faible corrélation entre les facteurs de production et la performance agronomique des systèmes. Le modèle STICS ne permet pas d'estimer les pertes de rendement associées aux stresses biotiques. Nous avons pu identifier 4 facteurs principaux ayant une influence significative sur les émissions d'azote réactif : la dose d'azote organique, le volume d'eau d'irrigation, le pH du sol et la capacité au champ. Nous avons également mis en évidence des incertitudes associées à la modélisation des processus de nitrification et de dénitrification.

Nous pouvons conclure que l'intégration de ces 4 facteurs influents des émissions d'azote réactif permet de prendre en compte une part significative de la variabilité des émissions au champ dans notre cas d'étude. Notre analyse montre que la collecte d'un certain nombre de données spécifiques relatives à la variété de tomate, au potentiel de dénitrification des sols et aux engrais organiques permettrait d'améliorer la qualité des sorties du modèle. Néanmoins les incertitudes issues de l'absence de prise en compte des pertes biotiques par le modèle STICS et issues du manque de jeux de données pour valider les processus de nitrification, de dénitrification, et de devenir de l'azote minéral du sol après récolte sont les principales limites pour l'utilisation de ce type d'approche pour estimer finement les émissions au champ dans notre contexte d'étude.

Mots-clefs : Bilan azoté · Modèle biophysique · Azote réactif · Variabilité · Incertitude

Closing the nitrogen budget for tomato cropping systems in African urban gardens

Prospects for Nr emission estimation in LCA

3.1. Introduction

Large quantities of nutrients in the form of chemical fertilizers and livestock manure are applied to urban gardens (Khai et al., 2007; Diogo et al., 2010; Sangare et al., 2012). Due to low yields, an important part of nitrogen input is lost to the environment (Predotova et al., 2010; Predotova et al., 2011; Lompo et al., 2012). Reactive nitrogen emissions (N_r emissions being NH_3, N_2O, NO_x and NO_3) often contribute considerably to the final results of the Life-Cycle Assessment (LCA) studies of vegetable products (chapter 1). However, it is often difficult to derive exact rates of N released to the air and to the water, because emission rates can greatly vary depending on soil type, climatic conditions and agricultural management practices. Brentrup et al. (2000) recorded three ways of taking these Nr emissions into account:

- To measure actual emission rates caused by the system under consideration,
- To use values derived from literature in a case-by-case procedure,
- To estimate potential emission rates using structured estimation methods.

The option of measuring emission rates directly on field is often discarded in LCA due to time and cost constraints as well as issues regarding variability of emissions at small scale (Bussink & Oenema, 1998; Stehfest & Bouwman, 2006). Deriving values from the literature requires the identification of representative references for the system. However, we found out that the best available references resulted in the estimation of emissions much different from measurements in similar systems (chapter 2, Table 2.3). Without better knowledge on drivers for emissions in our contexts we cannot rely on those measurements. Advantages of the last option (i.e. using structured estimation methods) are their easy performance, implying less effort compared to do site –specific measurements and often their better representativeness to the studied system compared to values derived from the literature (Brentrup et al., 2000).

According to Audsley et al. (1997) structured estimation method should be based on the calculation of mineral balance to ensure that the total input is equal to the total output and thus reduce the error on the more uncertain component of the budget, i.e. Nr emissions. In line with the LCA framework, spatial and time limits must be set to differentiate the system under analysis from its environment (SETAC, 1998). Due to short crop cycles and several crop successions within a year, temporal limits

on a year-basis were found unsuitable for vegetable crops (chapter 1). To be in line with the literature in agricultural sciences, the notion of "cropping system" (Sebillotte, 1990) can be applied as a set of management procedures applied to a given, uniformly treated agricultural area, setting the atmosphere and surrounding water pieces (groundwater and surface waters) as part of the environment. The nitrogen budget at cropping system scale records all nitrogen inputs and outputs, including nitrogen gains and losses within and from the crop-soil system.

Oenema et al. (2003) defined two main sources of uncertainty when calculating nutrient budgets: fundamental uncertainties related to the completeness and the reliability of the method to calculate budget components, and operational uncertainty, arising from the lack of data and knowledge, the variability in space and time, and the changes in items and parameters. Conditions which influence the nitrogen emissions are reflected by certain parameters (soil, climate, agricultural practice) which should be available and used as an input for the estimation methods (Brentrup et al., 2000). We found out that most up-to-date knowledge on Nr dynamics was embodied in modeling tools allowing the simulation of Nr emissions (chapter 1, section 1.4.4). Several LCA studies already used dynamic crop models to take into account local soil and climatic factors for the estimation of field-emissions, especially regarding energy crops and manure management where Nr emissions are crucial for the life cycle impact assessment (Adler et al., 2007; Gabrielle & Gagnaire, 2008; Langevin et al., 2010; Bessou et al., 2013; Dufossé et al., 2013). However, none of them had to face large uncertainty issues due to a context of extreme data scarcity associated with variable agricultural practices.

Based on the case study of tomato production in Beninese urban gardens, we applied the crop model STICS to simulate nitrogen fluxes at cropping system scale and complete a nitrogen budget at cropping system scale. Our objectives were: (1) identifying main influencing factors for nitrogen fluxes and emission factors, (2) comparing the nitrogen budgets from 12 tomato cropping systems, (3) identifying uncertainty issues and provide with recommendations for improving the reliability of Nr emissions estimates.

3.2. Materials and methods

3.2.1. An approach combining a crop model and a nitrogen budget

Selection of the crop model

Several crop models allows the simulation of Nr emissions such as NGAS-DAYCENT (Parton et al., 1996; Parton et al., 1998; Del Grosso et al., 2000), DNDC (Li et al., 1992; Li, 2000) or, CERES-NOE (Hénault et al., 2005; Gabrielle et al., 2006). The last one can be liken to the STICS model as it includes similar components for the simulation of gaseous emissions. Advantages of the STICS model are: (1) to be a generic and robust model based on mechanistic formalisms and able to simulate various soil and climate conditions with reasonable accuracy (Brisson et al., 2002), (2) to allow the simulation of tomato yields (Valdés-Gómez et al., 2009), (3) to be adaptable to tropical contexts (Sierra et al., 2003; Scopel et al., 2004; Sierra et al., 2010). STICS shown a fair performance in simulating carbon and nitrogen dynamics in arable soils (Gabrielle et al., 2002) and has already been used and to simulate nitrate leaching (Schnebelen et al., 2004; Jégo et al., 2008) and gaseous emissions (Langevin, 2010). Data required to run the model are accessible and relate to climate, cropping system (crop type and rotation, and agricultural practices) and soil properties (unchanging soil attributes and initial water and nitrogen profiles).

STICS simulates both agronomic variables (leaf area index, biomass, yield and input consumption) and environmental variables (soil profile water and contents, water drainage, nitrate leaching at the base of the soil profile, ammonia volatilized and nitrous oxide emissions). Figure 3.1 presents the conceptual model for the four output variables of interest for the case study. Nitrogen transformations and transfers simulated in STICS are: volatilization, mineralization and immobilization, nitrification, denitrification, leaching and crop exportation. Volatilization from mineral and organic fertilizers leads to NH_3 emissions. Nitrification ($NH_4 \rightarrow N_2O\uparrow \rightarrow NO_2 \rightarrow NO_3$) and denitrification ($NO_3 \rightarrow NO_2 \rightarrow NO \rightarrow N_2O\uparrow \rightarrow N_2\uparrow$) processes lead to N_2O, and eventually to N_2 emissions. Leaching lead to NO_3 emissions and yield formation leads to nitrogen exportations with harvested fruits. The formalisms responsible for Nr emissions will be further described in section 3.2.2. Within STICS the mineralization of soil organic matter is affected by soil relative water content (SrWC=soil water content/field capacity), as described in Rodrigo et al.(1997), the soil temperature as described in Valé (2006) and soil properties (bulk density, calcareous and clay content). STICS also simulates the mineralization of organic residues (i.e. organic fertilizers in our case) as described in Nicolardot et al. (2001) depending on organic fertilizers properties (see section 3.2.3). Both mineralization processes lead to nitrogen inputs to the system, while immobilization leads to nitrogen outputs. Finally, the nitrogen contained in exported fruits is calculated within the process of yield formation. This process is based on the phenology development influenced by the temperature and the biomass production influenced by the radiation, and is limited by stresses influenced by water and nitrogen availability.

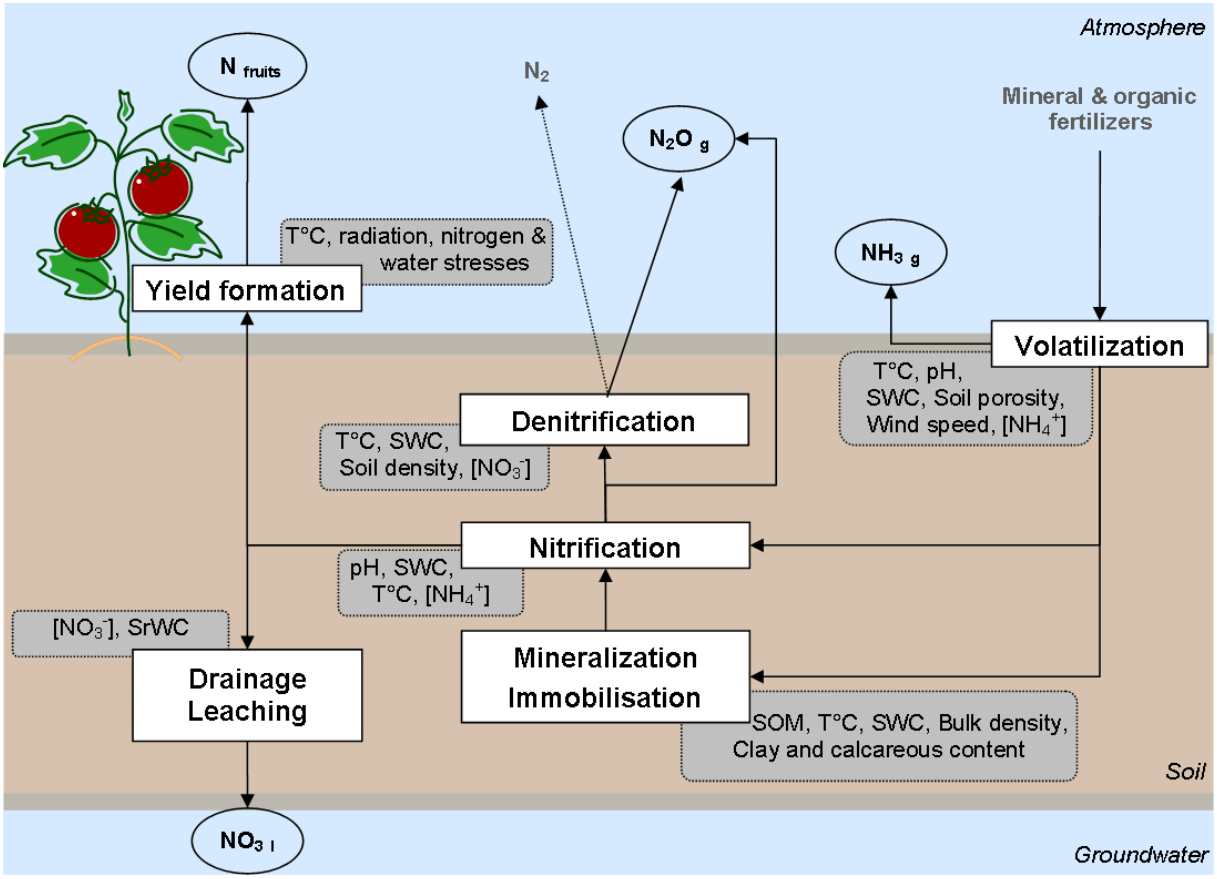


Figure 3.1 Conceptual model for output variables of interest (circles) as simulated in STICS: main influencing processes (white rectangle) and driving parameters (grey rectangles)

Calculation of the nitrogen Budget

Basically, an element balance equation for any system can be stated as (Oenema et al., 2003):

$$\Delta P_E = I_E - O_E \qquad \textbf{Eq. 1}$$

where ΔP_E, I_E and O_E stand for the change in the pool, the input and the output of element E, respectively. Khai et al. (2007) applied Eq. (1) for the field balance of nutrients in an agricultural system. The input flows were derived from seedlings (S_E), irrigation water (IW_E), rainwater (RW_E), fertilizers (F_E), atmospheric deposition (AD_E), symbiotic nitrogen fixation (N_E), the output flows from harvested organs (H_E), leaching (L_E), runoff (R_E), air emission (AE_E), and ΔP_N was the net change in soil storage of mineral nitrogen ($\Delta Soil_E$). Thus, Eq. (1) can be modified to:

$$\Delta Soil_E = S_E + IW_E + RW_E + F_E + AD_E + SF_E - (H_E + L_E + R_E + AE_E) \qquad \textbf{Eq. 2}$$

Soil nitrogen budgets were calculated over the cropping system period using Eq. 2 proposed by Khai et al. (2007). Applying Eq. (2) using variables simulated by the crop model STICS (Figure 3.1), we first had to include input from mineralization of soil organic matters (MH_N), output from immobilization into soil organic matter (IH_N). We also detail nitrogen fertilizers as organic fertilizers

(FO_N) and mineral fertilizers (FM_N). Then we refined atmospheric emissions (AE_N, eq. 3) as ammonia from volatilization of organic (VO_N) and mineral (VM_N) fertilizers, nitrous oxide from nitrification (Nit_N) and denitrification ($Denit_N$) and, dinitrogen from denitrification ($Denit_{N2}$). Even if N_2 is not of environmental relevance, N_2 rates emitted to the air should be included in the nitrogen balance. Indeed, it is the main product of denitrification and usually released in much higher rates than N_2O. NO_x emitted through nitrification (Nit_{NO}) was the only Nr emission which was not simulated by the crop model STICS. However STICS simulated the processes of nitrification and we can estimated on average that 0.5% of nitrified nitrogen is emitted as NO_x (Rolland et al., 2010).

$$AE_N = Denit_{N2O} + Denit_{N2} + Nit_{N2O} + Nit_{NO} + VO_{NH3} + VM_{NH3} \quad \textbf{Eq. 3}$$

Finally, considering the high levels of nitrogen fertilizers rates and in agreement with several studies dealing with nitrogen budgets of urban gardens in Africa (Lompo, 2012; Sangare et al., 2012) we set nitrogen amounts from seedlings (S_E), irrigation water (IW_E), rainwater (RW_E) and atmospheric deposition (AD_E) to zero. Thus Eq. (2) can be modified to Eq. (4) with inputs at the left hand and output at the right hand:

$$FO_N + FM_N + MH_N =$$
$$IH_N + H_N + Denit_{N2O} + Den_{N2} + Nit_{N2O} + Nit_{NO} + VO_{NH3} + VM_{NH3} + L_{NO3} + \Delta Soil_N \quad \textbf{Eq. 4}$$

3.2.2. Calculation of Nr emissions in STICS

This section is limited to a description of the STICS model (version 11) for specific variables of interests within this study. The reader can refer to Brisson et al. (2008) for more information about conceptual basis, formalizations and parameterization.

Volatilization

The volatilization of ammonia (NH_3) from nitrogen-based fertilizers is divided into two processes calculated on a daily-step basis (d): volatilization from mineral fertilizers and from organic residues. First, the volatilization rate from mineral fertilizers (VM_N, eq.5) depends on the crop nitrogen uptake rate (VABSMOY), the amount of mineral nitrogen applied (FM) and soil pH (FPH) according to the following relationship:

$$VM_{NH3}(d) = VOLENG_P(f) \times \frac{VABS2_P}{VABS2_P + VABSMOY(I)} \times FM(d) \times FPH \quad \textbf{Eq. 5}$$

with $VOLENG_P$ a fertilizer specific parameter defining the maximal proportion of nitrogen losses by volatilization (see section 3.2.3) and $VABS2_P$ a parameter defining the crop uptake rate at which volatilization losses reach 50% of their maximum (2 $kg.ha^{-1}.d^{-1}$). Figure 3.2 presents the variations of the pH factor according to soil pH.

Second, the volatilization rate from organic fertilizers (VO_N, eq. 6) depends on the potential NH_3 volatilization rate (FSNH3, in µg N.m-2.s-2):

$$VO_{NH3}(d) = 0.036 \times FSNH3\ (d) \qquad \textbf{Eq. 6}$$

The calculation of FSNH3 is based on equations developed by Génermont and Cellier (1997) for the acido-basic and liquid-gas equilibria and varies according to the concentration of ammonium (NH_4) in the top-soil, CEC, soil water content, pH, temperature and wind speed (Beutier & Renon, 1978; Shuttleworth & Wallace, 1985). The volatilization from organic fertilizers (VO) cannot exceed the amount of NH_4 at soil surface nor the NH_4 volatilizable pool. On the day of application, the volatilizable pool corresponds to the NH_4 content of the organic fertilizer minus the fraction which infiltrates into the subsoil. This volatilizable fraction, so called PROPVOLAT, depends on the dry matter content of the organic fertilizer and is assumed to be affected by soil tillage (Figure 3.3). The volatilizable pool is then updated daily.

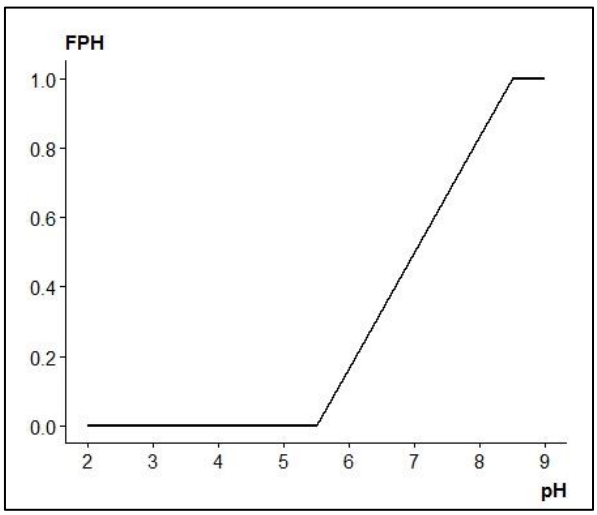


Figure 3.2 Effect of soil pH on volatilization from mineral fertilizers

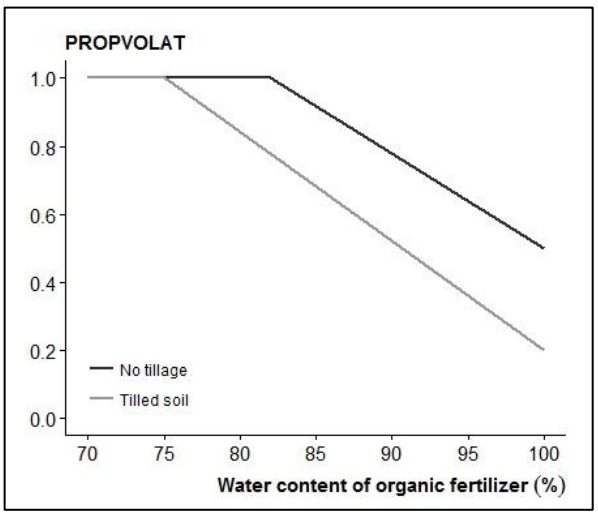


Figure 3.3 Effect of dry matter content of fertilizer and soil tillage on the volatilizable fraction

Nitrification and denitrification

In STICS, the emissions of nitrous oxide (N_2O) result from two main processes calculated on a daily-step basis: nitrification and denitrification. N_2O emitted through nitrification (Nit_N, eq. 7) depends first on the fraction of NH_4 transformed into nitrate (NO_3) in each layer (TNITRIF, eq.8) from the surface to the last active layer for mineralization (PROFHUM). It also depends on the amount of NH_4 in the specific soil layer (Z). N_2O emitted is assumed to be a constant proportion of the nitrified NH_4 (Garrido et al., 2002), so called $RATIONIT_P$ (with a default value of 0.0016):

$$Nit_{N2O}(d) = RATIONIT_P \sum_{z=1}^{PROFHUM} TNITRIF\ (Z,d) \times NH_4\ (Z,d) \qquad \textbf{Eq. 7}$$

The fraction of NH_4 transformed into NO_3 (TNITRIF, eq. 10) cannot exceed the daily maximum fraction of ammonium transformed into nitrate (FNX_P = 0.5, Sierra et al. (2003)) and is affected by soil pH (FPHN), soil relative water content (FHN) and soil temperature (FTN):

$$TNITRIF\ (Z,d) = FNX_P \times FPHN \times FHN\ (Z,d) \times FTN\ (Z,d) \quad \textbf{Eq. 8}$$

Figure 3.4 presents the variations of the pH factor, Soil relative water content factor (SrWC) and the temperature factor.

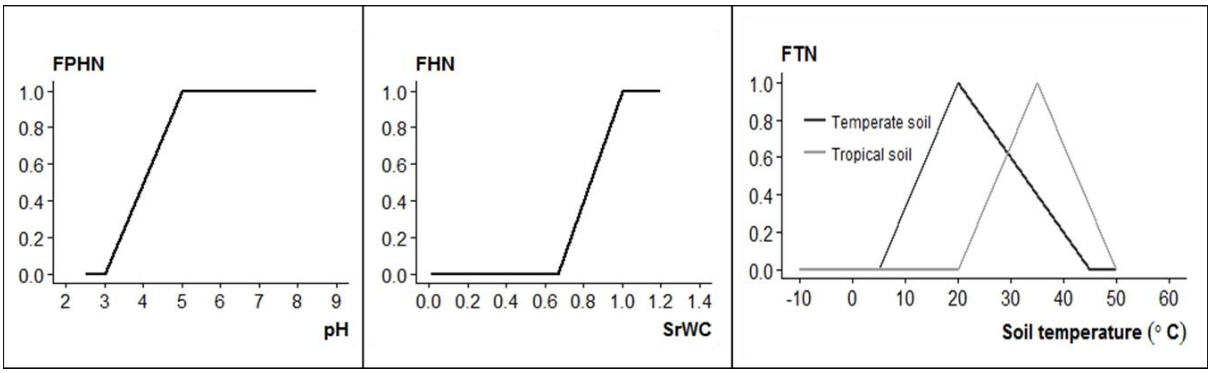


Figure 3.4 Effects of pH (FPHN), Soil relative water content (FHN) and soil temperature (FTN) on nitrification

At this stage, we can also deduce quantities of NO_x emitted through nitrification (Nit_{NO}, eq. 9) based on measurements from Rolland et al. (2010):

$$Nit_{NO}\ (d) = 0.005 \times \sum_{z=1}^{PROFHUM} TNITRIF\ (Z,d) \times NH_4\ (Z,d) \quad \textbf{Eq. 9}$$

N_2O emitted through denitrification ($Denit_N$, eq. 10) is calculated according to the model proposed by Hénault et al.(2005). It is a constant proportion of the denitrified NO_3, so called $RATIODENIT_P$ (with a default value of 0.1). For each denitrifying layer ($PROFDENIT_P$, 0-20cm) N_2O losses cannot exceed the total denitrification potential rate (1kg $N.ha^{-1}.d^{-1}$) and are affected by soil temperature (FDENT), the amount of nitrate in soil (FDENNO3) and soil properties (FDENW):

$$Denit_{N2O}\ (d) = RATIODENIT_P \times \frac{VPOTDENIT_P}{PROFDENIT_P} \times$$
$$\sum_{z=1}^{PROFDENIT_S} FDENT\ (Z,I) \times FDENNO3\ (Z,d) \times FDENW\ (Z,I) \quad \textbf{Eq. 10}$$

Figure 3.5 presents the variations of the soil temperature factor, the soil nitrate content factor and the soil properties factor referring to both water content at field capacity (FC) and bulk density (BD).

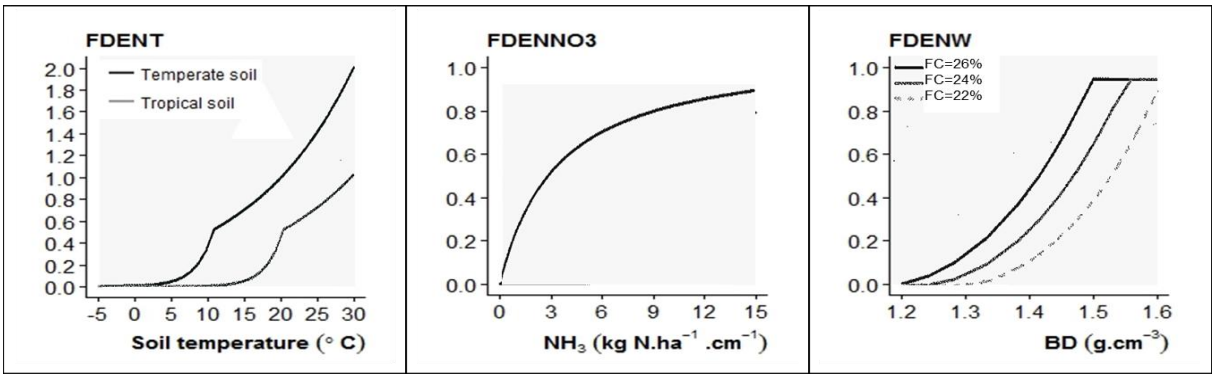


Figure 3.5 Effects of soil temperature (FDENT), soil nitrate content (FDENNO3) and soil properties (FDENW) on denitrification

At this stage, we can also deduce quantities of N_2 emitted through denitrification ($Denit_{N2}$, eq. 11):

$$Denit_{N2}\,(d) = (1\text{-}RATIODENIT_P) \times \frac{VPOTDENIT_P}{PROFDENIT_P} \times \sum_{z=1}^{PROFDENIT_S} FDENT\,(Z,d) \times FDENNO3\,(Z,d) \times FDENW\,(Z,d) \qquad \textbf{Eq. 11}$$

Nitrate leaching

Water transfers are calculated using a reservoir-type analogy. The excess water above field capacity is drained downward and leads to leaching of nitrates (NO_3) soluble in the drained soil water.

3.2.3. Parameterization of the model for the case study

Data for agricultural practices were collected for 12 cropping systems representative of the production of out-of-season tomato in Beninese urban gardens. This section is limited to a brief description of the data used to feed the model for this case study. The reader can refer to the previous chapter for more detail on the data collection procedure and inventory data for these cropping systems. The data refers to the 2011/2012 dry season (i.e. from September to April).

STICS input consists in four files: climate, soil, management and initialization. The climate file contains daily data regarding rainfall (mm), wind speed ($m.s^{-1}$), radiation ($MJ.m^{-2}$), minimum and maximum temperature (°C) and, relative humidity (%). The same file based on data from the international research center located at Cotonou (AfricaRice) was used for the simulation of the 12 cropping systems. The evapotranspiration (ETP) was automatically calculated in STICS based on the Shuttleworth and Wallace resistive approach (Shuttleworth & Wallace, 1985). Figure 3.6 shows variations of minimum temperature (CV=7%), radiation (CV=21%) and, ETP (CV=34%) were low. However, rainfalls were particularly low from December to March (CV=356%) and maximal temperature slowly increased from September to March (CV=69%).

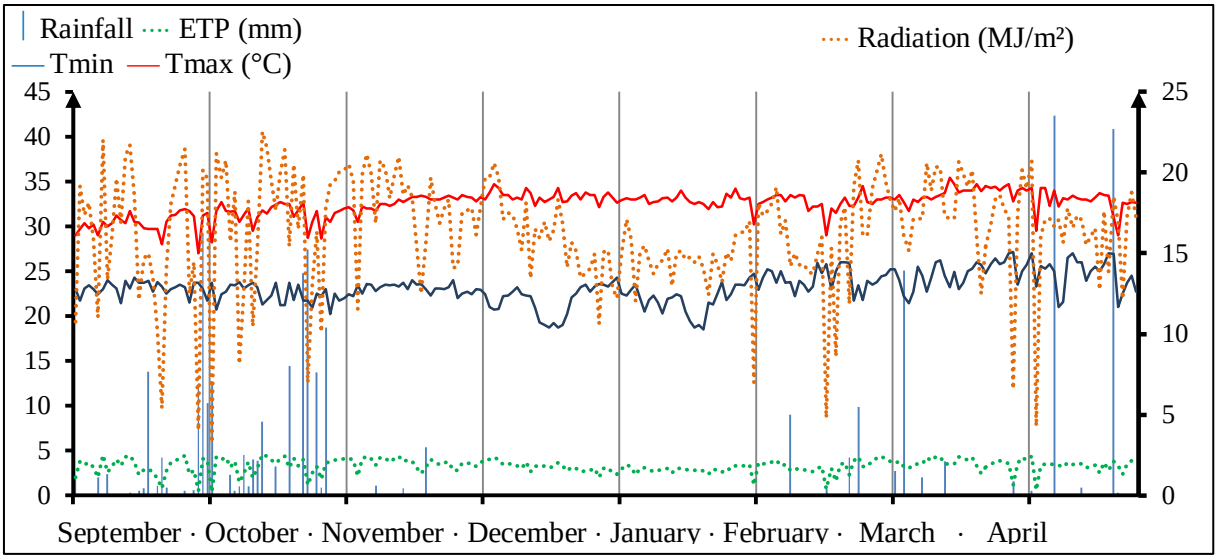


Figure 3.6 Summary of climate variables for the 2011/2012 dry season at Cotonou, Benin, West Africa

For each cropping system, the soil files contain two types of data due to data availability constraints: site specific data and estimations from the literature. Soil analyses were performed by the soil laboratory at the University of Abomey-Calavi from soil samples collected in the 12 farms at the beginning of each crop cycle. Table 3.2 presents site specific parameters, regarding clay, organic nitrogen and calcareous (Ca) contents, pH, C:N (carbon to nitrogen ratio), field capacity at saturation (FC_S) and at wilting point (FC_W) and bulk density (BD). Other parameters were estimated based on references from the model documentation(Brisson et al., 2008).

Table 3.2 Site-specific soil data used to feed the STICS model from 12 tomato cropping systems in Benin

	Clay	**Organic nitrogen**	**Total $CaCO_3$**	**pH**	**C:N**	**Moisture (%)**		**Bulk density**
	%	*%*	*%*	-	-	*Maximal capacity*	*Wilting point*	*$g.cm^{-3}$*
1	7.8	0.05	0.18	6.8	15.2	13.4	2.8	1.4
2	1.8	0.04	0.16	6.0	20.9	13.4	2.7	1.2
3	4.5	0.03	0.34	6.6	16.8	10.8	2.0	1.5
4	13.0	0.03	0.46	5.7	30.4	13.6	5.2	1.2
5	9.8	0.05	0.34	6.0	21.8	16.1	4.0	1.3
6	15.0	0.05	0.18	6.8	19.0	10.6	3.1	1.4
7	0.8	0.02	0.57	6.4	23.5	9.6	1.7	1.5
8	3.3	0.03	0.34	6.9	19.1	11.4	2.2	1.2
9	2.0	0.06	0.34	6.6	17.9	18.6	3.4	1.5
10	2.0	0.03	0.57	4.8	21.8	13.0	3.5	1.5
11	7.8	0.05	0.23	5.0	22.2	24.7	4.2	1.4
12	65.0	0.13	0.11	4.6	20.6	61.6	33.3	1.3

For each cropping system, the management files contain data regarding transplanting (date, crop density), irrigation (date and volumes for each irrigation event), fertilization (date and rates for each

fertilization event) and harvest (harvest frequency, last harvest date). Table 3.3 presents those management data. For each type of mineral fertilizer (NPK vs. Urea), a set of parameters was defined based on the STICS documentation:

- Ammonium proportion (100 % vs. 50 %),
- Maximal proportion of N losses by volatilization (35% vs. 12%),
- Maximal amount of microbial immobilized N (37.5 kg $N.ha^{-1}$ vs. 30.0 kg $N.ha^{-1}$)

For poultry manure the set of parameters was taken from measurements from Chabalier et al. (2006):

- Carbon content (22%)
- C:N ratio (6)
- Mineral nitrogen content (3%)
- Water content (54%)

Table 3.3 Management data used to feed the STICS model from 12 tomato cropping systems in Benin

	Length	**Yield**	**Transplant**	**Crop density**	**Water**	**Poultry manure**	**Urea**	**NPK**	**Last harvest**
	days	*kg.ha^{-1}*	*date*	*plt.m^{-2}*	*m^3.ha^{-1}*	*kg N.ha^{-1}*	*kg N.ha^{-1}*	*kg N.ha^{-1}*	*date*
1	80	4902	9-Oct	3.9	7200	1045	33	107	28-Dec
2	64	12452	11-Nov	5.3	6360	64	146	104	14-Jan
3	80	8	5-Oct	1.8	586	117	0	19	24-Dec
4	48	7875	4-Nov	2.7	10400	473	43	25	22-Dec
5	92	21163	30-Oct	7.4	5318	0	0	22	30-Jan
6	82	127	21-Nov	3.8	4515	563	0	235	11-Feb
7	36	0	21-Nov	2.4	919	0	0	58	27-Dec
8	91	5662	14-Dec	3.3	10581	778	0	0	14-Mar
9	83	1703	5-Dec	5.7	7048	0	59	0	26-Feb
10	79	4498	5-Jan	3.3	771	0	202	72	24-Mar
11	57	2710	17-Nov	3.5	3759	450	0	0	13-Jan
12	67	0	23-Nov	3.7	2570	559	0	211	29-Jan

In the initial condition file, the nitrogen content was set to 0 and the water content was set to 100% of the field capacity, as important water volumes were applied to tomato before transplanting, leading to the leaching of most of the residual soil nitrogen.

STICS also requires a set of general parameters regarding the crop and the formalisms. Due to a lack of data specific to our crop growth, we used the generic tomato crop file, with the Capitan cultivar as the better approximation for our determinate tropical cultivar (Mongal). We did not change general parameters except for climate dependent ones (Table 3.4).

Table 3.4 General parameters used to adapt the STICS crop model to tropical conditions

Process involved	Parameter description	Temperate	Tropical
N transfer	Threshold of soil nitrate concentration for leaching (kg N.ha^{-1})	1.01 (1)	0.20 (1)
Mineralization	Minimal water content for mineralization (% of field capacity)	0.30 (2)	0.22 (1,3)
	Optimal water content for mineralization (% of field capacity)	1.0 (2)	0.67 (1,3)
Nitrification	Minimal temperature below which nitrification stops	5 (1)	20 (1)
	Optimal temperature for nitrification	20 (1)	35 (1)
	Maximal temperature above which nitrification stops	45 (1)	50 (1)
Denitrification	Parameters for the effect of temperature	11 (1)	20 (4)
		20 (1)	29 (4)

1. Brisson et al. (2008) **2.** Rodrigo et al. (1997) **3.** Sierra et al. (2003) **4.** Sierra et al.(2010)

3.2.4. Statistical analysis

Variations of observed and simulated variables were analyzed using simple and multivariate linear regressions computed with the R software (V 2.15.2). We thus assumed that the dependent variable (*Y)* was a linear combination of the parameters (*a* and *b*) and the explanatory variables (X, i.e. Y=*a*X+*b*). The coefficient of correlation (r^2) corresponds to the fraction of variance explained and the slope (a) informs about the effect of the predictor on the dependent variable. The significance of the slope (a) was tested using a Student's t-test with H_0: a=0 and H_1:a≠0. Significance codes were assigned (`for 5%,* for 1%, ** for 0.1% and *** for less than 0.1%) corresponding to the risk in accepting H_1 when H_0 is true. If the slope was not significant, we could not conclude on an effect of the predictor. When necessary, the independence of variables was tested with a Pearson's chi-squared test where the contingency coefficient (CC) is a measure of the degree of relationship between the dependent variables.

3.3. Results and discussion

3.3.1. Quality of prediction and model soundness for yields

On a crop cycle basis, STICS consistently overestimated yields except for one cropping system (Figure 3.7). As a result we ended up with a poor quality of prediction with r^2=0.25. The distribution of the error seems to follow a linear function (Figure 3.8, R^2=0.51) indicating particularly poor predictions for low yields.

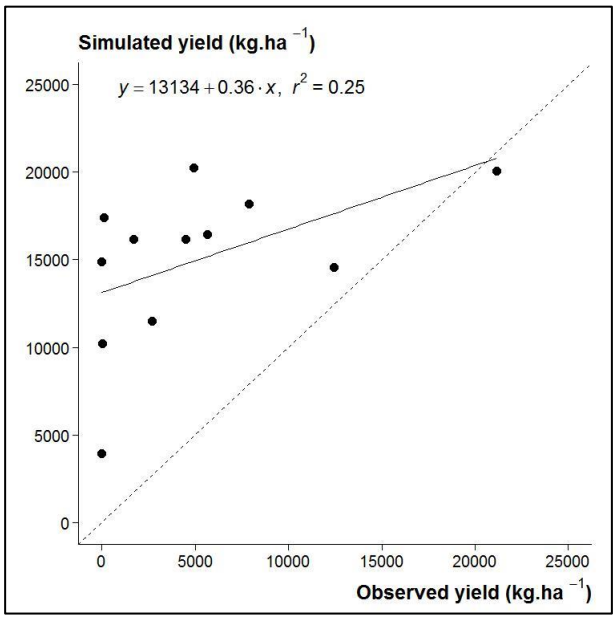


Figure 3.7 Simulated versus observed yields

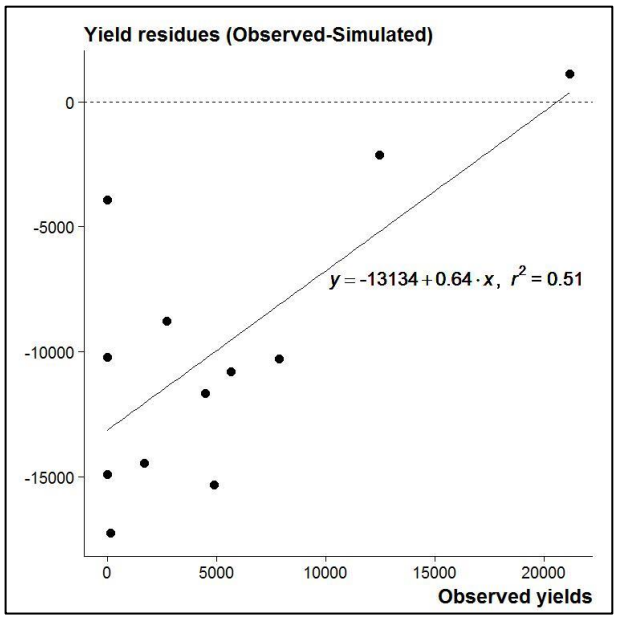


Figure 3.8 Yield residues versus observed yields

To analyze the discrepancy between simulated and observed yields, we tested the effect of identified drivers for the process of yield formation within STICS (Figure 3.1) on simulated and observed yields. Figures 3.9 and 3.10 show that an important part of simulated yield variations can be explained by the cumulative global radiation (r^2=0.65) and the cumulative temperature (r^2=0.66). It means that simulated yields were limited by climatic drivers (a=14** and a=67** for respectively radiation and temperature). The cumulated temperature being significantly correlated to the cumulative radiation (CC=0.97), we could not identify which one of these drivers was the most limiting for yields. The positive effect of total water, i.e. irrigation water plus rainfall (Figure 3.11) indicated that simulated yields were limited by water stresses (r^2=0.39*). However, yield and nitrogen inputs were correlated neither in the simulation nor from observations (Figure 3.12). The lack of correlation might result from the absence of nitrogen stresses resulting from excessive inputs. It might also results from important variations of nitrogen efficiency caused by important nitrogen losses.

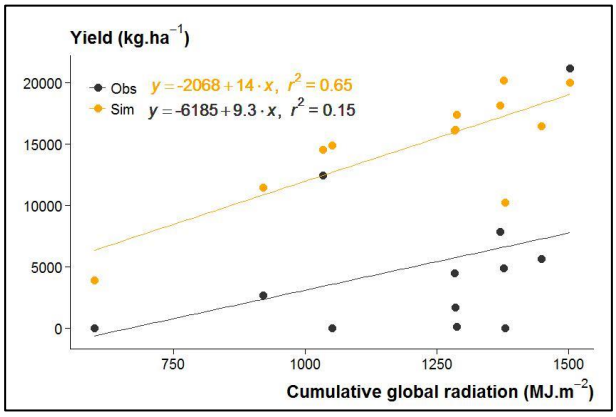


Figure 3.9 Simulated and observed yields versus cumulative global radiation over the crop cycle

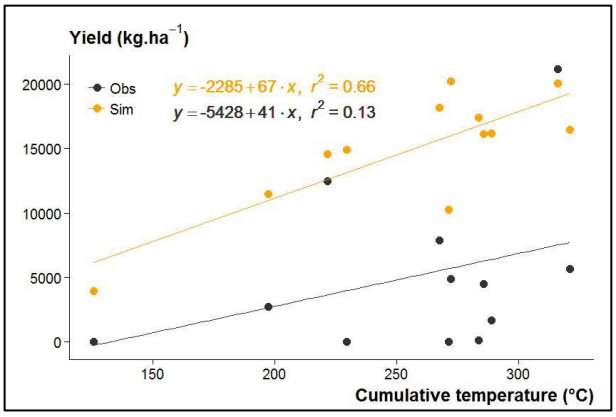


Figure 3.10 Simulated and observed yields versus cumulative temperature over the crop cycle

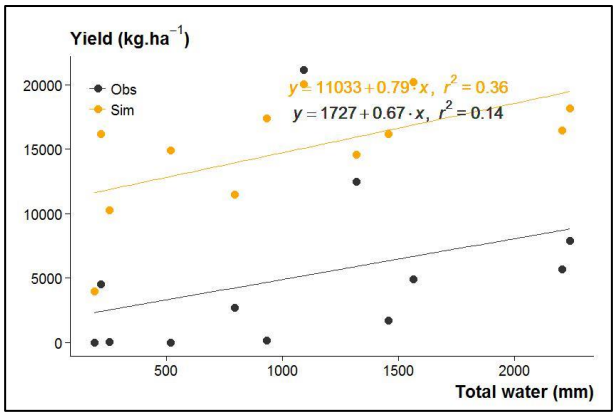


Figure 3.11 Simulated and observed yields versus total water

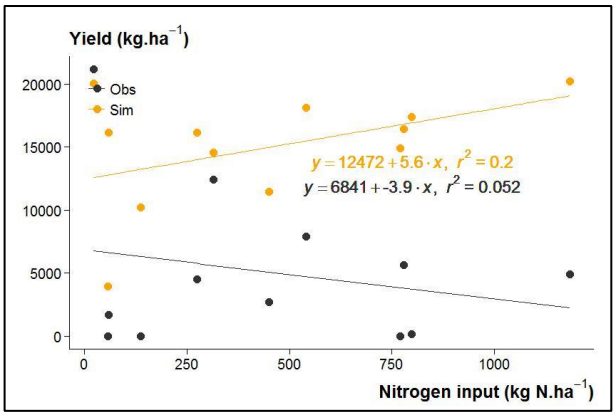


Figure 3.12 Simulated and observed yields versus nitrogen inputs

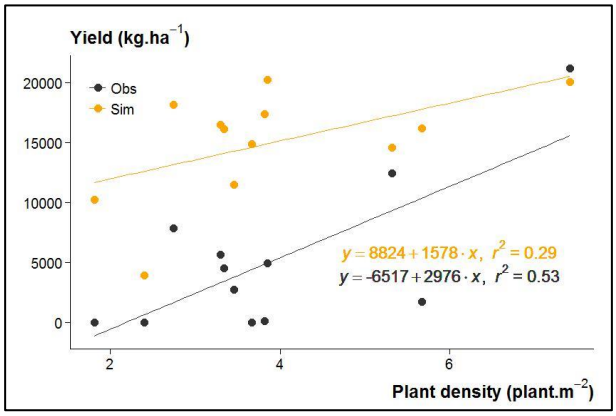


Figure 3.13 Simulated and observed yields versus plant density

The weak correlation between observed yields and those drivers (radiation, temperature, nitrogen, water) indicated that other drivers were more limiting for observed yields. Figure 3.13 shows that plant density was the best explicative variable we found for observed yields (r^2=0.53). The higher effect of plant density for the observed yield (a=2976**) compared to simulated yields (a=1578`) suggested a higher production per plant on field than the one simulated by STICS.

We could not properly assess the model efficiency to simulate yields because we could not compare simulated yield to yield obtained in controlled conditions. The extreme biotic pressure on studied cropping systems resulted in major yield variations which could not be reproduced with STICS (maximum r^2 was 0.53). However we could test the model soundness. We can conclude that simulated tomato yields were sensitive to climate conditions (temperature and/or radiation) and volumes of water applied. Considering the dry and hot climate and the variable water volumes applied across fields, STICS behavior can be considered as satisfactory. For these 3 drivers (radiation, temperature and water), effects were similar for simulated and observed yields (same slope). It suggests that STICS succeeded in simulating the response of yields to these drivers. However, the low correlation of observed yield to these drivers indicated that observed yield were mainly affected by other drivers. In agreement with Huat et al. (2013) who identified planting density and pest attacks as main limiting drivers for yields of tomatoes fields grown by smallholders in tropical regions, a significant part of yield variations could be explained by plant density whether in the simulations or in the observations. The higher effect of plant density for observed yield suggested that parameters affecting the production per plant were not adapted to our tropical cultivar.

3.3.2. Analysis of drivers for nitrogen losses as simulated by STICS

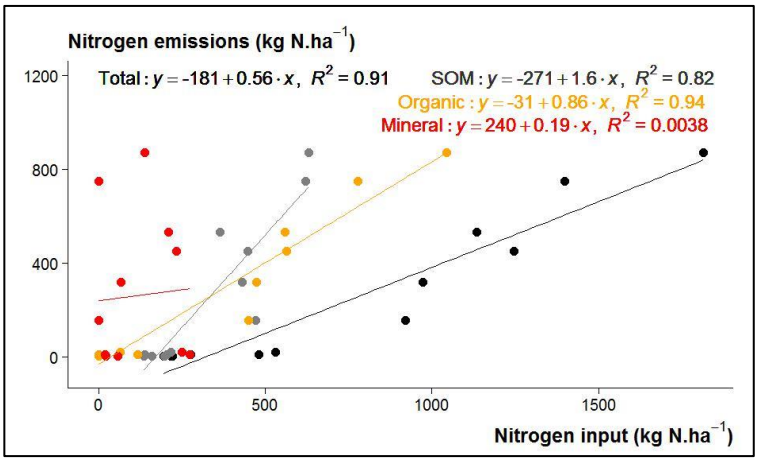


Figure 3.14 Total Nr emissions versus nitrogen input from mineralization of soil organic matter, organic and mineral fertilizers

In Life-cycle assessment, Nr losses (NH_3, N_2O, NO_x and NO_3) to the environment are estimated using emission factors, i.e. the ratio between the amounts of nitrogen emitted on the amount of nitrogen input. Figure 3.14 shows a correlation between the amounts of nitrogen emitted on the amount of nitrogen input (r^2=0.91). It appears that main drivers for nitrogen emissions are organic fertilizers

inputs (r^2=0.94) followed by nitrogen from soil organic matter mineralization (r^2=0.82). In the following section we will analyze the sensitivity of emissions factors for NH_3, N_2O and NO_3 to soil, management and climate drivers.

Ammonia volatilization

The total ammonia emission factor (NH_3 emissions/total fertilizer inputs) varied from 0 to 37 % across the 12 cropping systems, with an average of 10%. These results were lower than NH_3 emissions factors (39% on average, see chapter 2 Table 2.3) calculated from ammonia measured on urban gardens in Niger (Predotova et al., 2010) and Burkina Faso (Lompo et al., 2012). NH_3 emission factors were expressed for each type of fertilizers, i.e. organic and mineral. The emission factors from organic fertilizers, corresponding to the main origin of emission, varied from 0.9 to 73% across the 12 cropping systems with an average of 31%. This result was higher to the emission factor of 20% suggested by Bouwman and Van Der Hoek (1997). We tested the correlation between NH_3 emission factors and the identified drivers for volatilization within STICS (Figure 3.1). We found out that NH_3 emission factors were correlated to soil pH (Figure 3.15). The effect of pH was more important for NH_3 emission from organic fertilizers (a=0.31**) than from mineral fertilizers (a=0.02**). The equations from Génermont and Cellier (1997) included an acido-basic equilibrium explaining the increase of emissions with soil pH through its effect on the NH_4^+:NH_3 ratio in the soil (Figure 3.2). However, we could not identify any other variable with a significant effect on NH_3 emissions from organic fertilizers. Regarding NH_3 emissions from mineral fertilizers, the volatilization rate was expected to increase with soil pH (equation 5). The second effect was due to the nature of the fertilizers: NPK or urea ($VOLENG_P$ for urea> $VOLENG_P$ for NPK). None of these drivers showed significant effect on NH_3 emissions from mineral fertilizers. Unexpectedly, NH_3 emissions from mineral fertilizers were correlated to the total water inputs (Figure 3.16). It suggests that the plant nitrogen uptake velocity (being the last effect variable in equation 5) was influenced by water inputs.

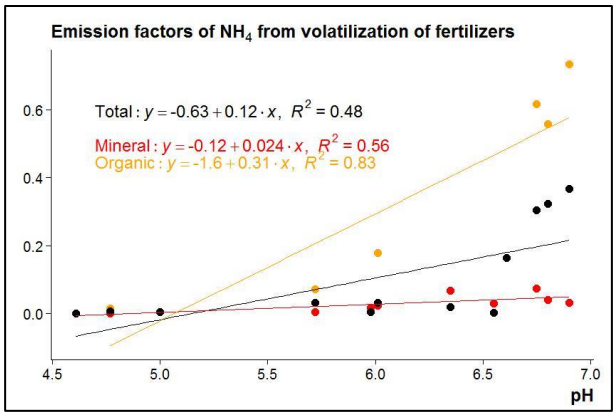


Figure 3.15 NH_4 emissions factors versus soil pH

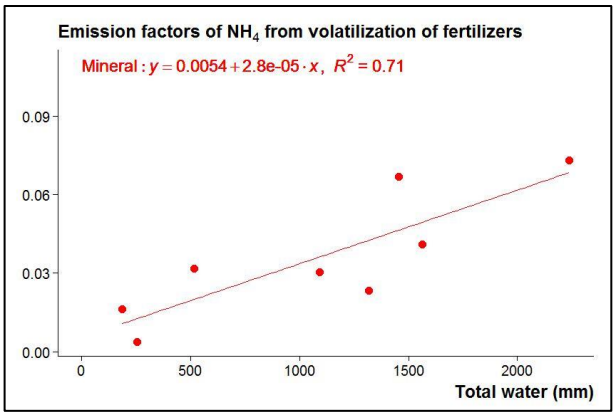


Figure 3.16 NH_4 emissions factors for mineral fertilizers versus total water

N_2O emissions from nitrification and denitrification

The total nitrous oxide emission factor (N_2O emissions /total nitrogen inputs including soil mineralization) varied from 0.08 to 1.96 % across the 12 cropping systems with an average of 0.28%. This result was lower than N_2O emissions factors (15% on average, see chapter 2, Table 2.3) calculated from N_2O measured on urban gardens in Niger (Predotova et al., 2010) and Burkina Faso (Lompo et al., 2012). N_2O emission factors were expressed for each process, i.e. nitrification and denitrification. We tested the correlation between N_2O emission factors and the identified drivers for nitrification and denitrification within STICS (Figure 3.1). The major part of N_2O emission factor from denitrification variations can be explained by soil moisture at maximal field capacity (r^2=0.93, Figure 3.17) even if we excluded the extreme clay soil (r^2=0.63, Figure 3.18). An important part of N_2O emission factor from nitrification variations, can be explained by soil pH (r^2=0.40, Figure 3.19). While the effect of changing pH from 4.5 to 5.5 agreed with the expected trend (Figure 3.2), the decrease of N_2O emission factor for great pH probably resulted from another influencing driver.

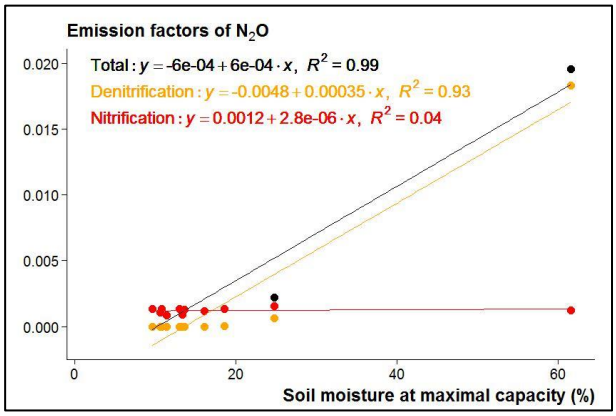


Figure 3.17 N_2O emission factors versus soil moisture at maximal capacity for all cropping systems

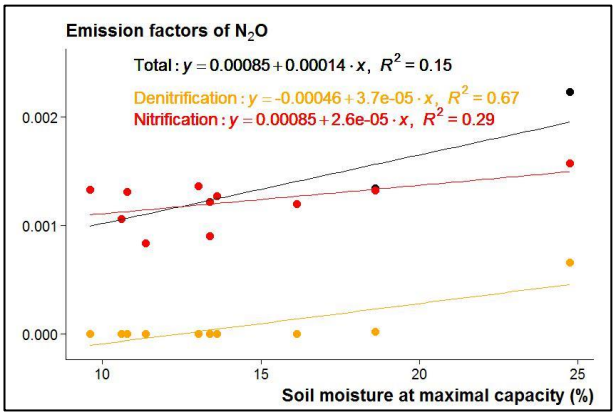


Figure 3.18 N_2O emission factors versus soil moisture at maximal capacity for sandy soils

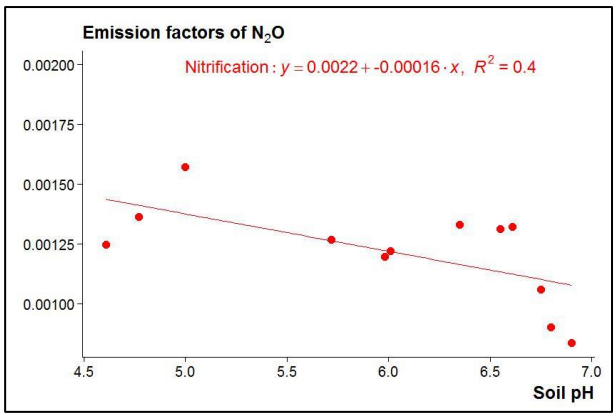


Figure 3.19 N_2O emission factors for nitrification versus soil pH

Nitrate leaching

The nitrate emission factor (NO_3 emissions/total nitrogen inputs including soil mineralization) varied from 0 to 52% across the 12 cropping systems, with an average of 10%. This result was higher than NO_3 emissions factors (1% on average, see chapter 2, Table 2.3) calculated from nitrate measured on urban gardens in Niger (Predotova et al., 2011) and Burkina Faso (Lompo, 2012), which is logical considering the intensive irrigation in our systems. Figure 3.20 shows that NO_3 emissions were positively correlated to the nitrification rate of (r^2=0.74, a**). This effect was in agreement with the reservoir-type approach for leaching, being the result of drainage of water containing NO_3. However, figure 3.21 shows a poor correlation between NO_3 emissions and the total water inputs (r^2=0.19). It suggests that other drivers influenced drainage events (probably a combination of soil parameters).

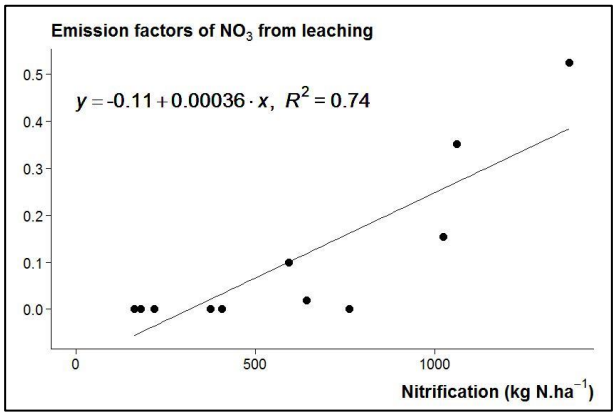


Figure 3.20 NO_3 emissions versus the total nitrified nitrogen

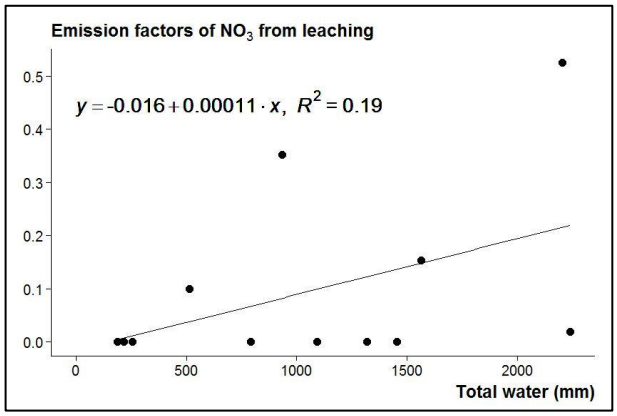


Figure 3.21 NO_3 emissions versus total volume of water

3.3.3. Evaluation of the combined approach

Figure 3.22 presents the nitrogen budgets for the 12 cropping systems simulated with STICS. Nitrogen budget for the 12 cropping systems differed in terms of importance (195 to 1814 kg N.ha^{-1}) and diversity of nitrogen fluxes.

The net change in soil storage of mineral nitrogen, also called nitrogen surplus ($\Delta Soil_N$) varied from 145 to 876 kg N.ha^{-1} with an average of 463 kg N.ha^{-1}. It represented on average 59% of the total output. Such a large nitrogen surplus agrees with results from several studies dealing with nitrogen budget of urban gardens (Diogo et al., 2010; Lompo, 2012). Nitrogen fertilizers constituted on average 57% of nitrogen inputs, with respectively 43% for organic fertilizers and 14% for mineral fertilizers.

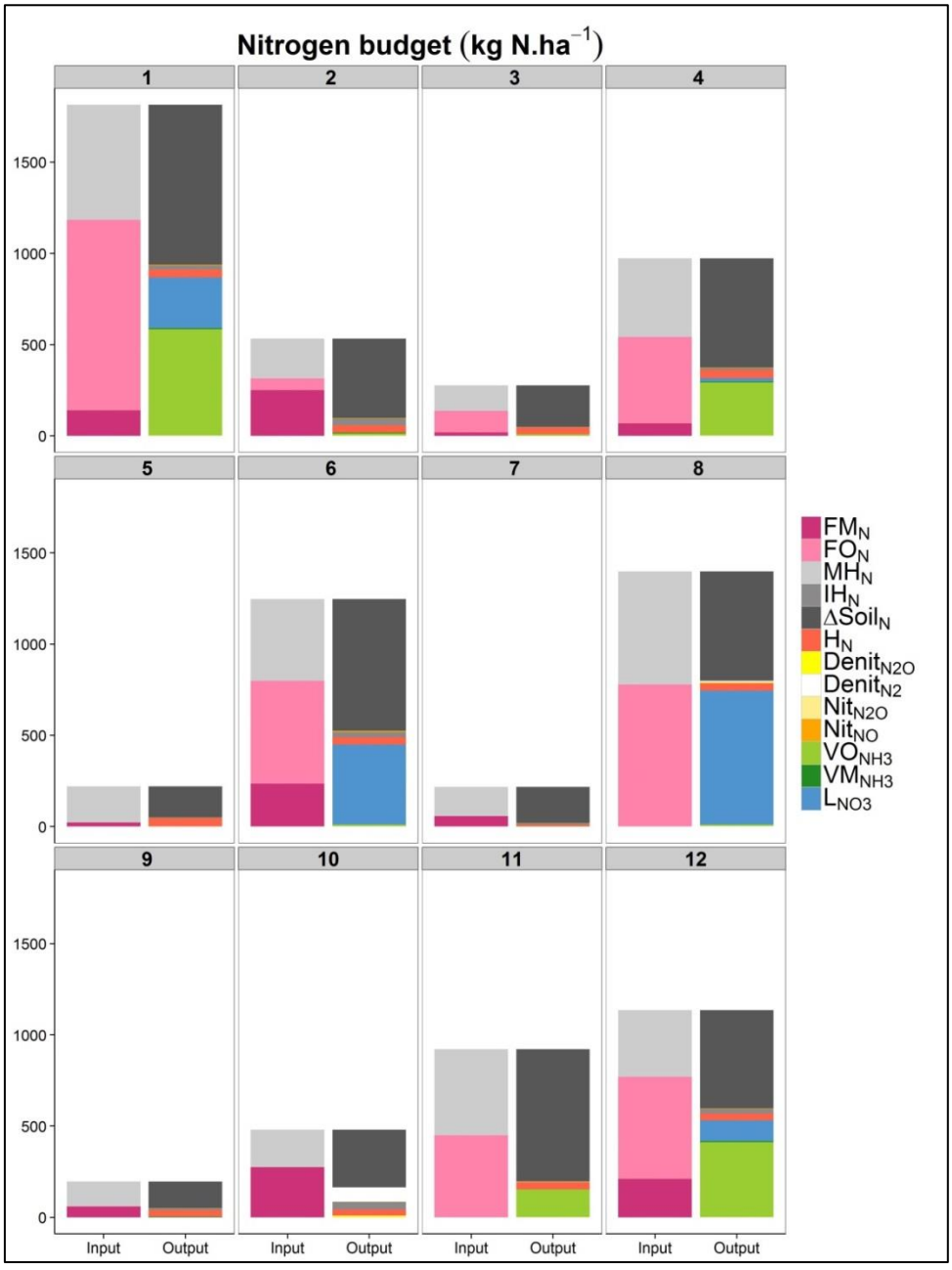


Figure 3.22 Nitrogen budget simulated with STICS for the 12 cropping systems

Soil mineralization

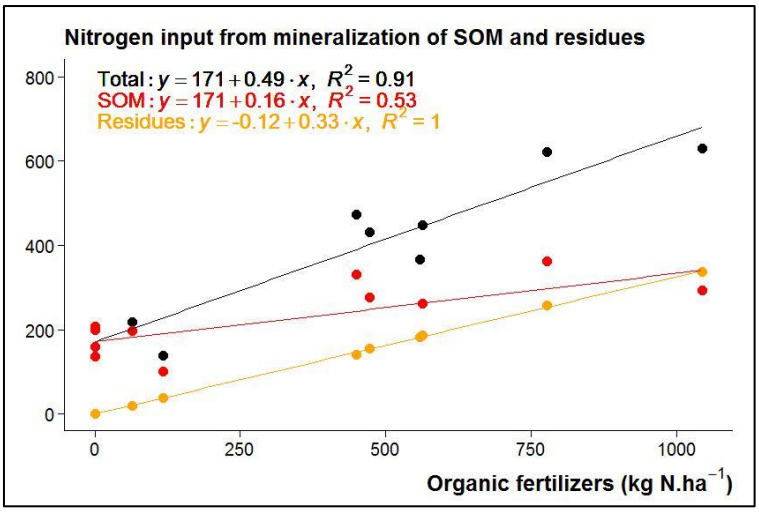


Figure 3.23 Nitrogen input from soil mineralization versus organic fertilizers

One particularity of using a dynamic crop model to calculate a nitrogen budget referred to the inclusion of the mineralization of soil organic matter and organic residues. While immobilized nitrogen into soil organic matter did not contribute much to nitrogen outputs (3% on average), nitrogen mineralized (MH_N) contributed on average 51% of nitrogen input, with variable quantities across cropping systems (CV=54%). Figure 3.23 shows that most of the variation of nitrogen from mineralization can be explained by organic fertilizers inputs (r^2=0.91). As expected, the nitrogen mineralized from residues (i.e. from organic inputs) was perfectly correlated to organic fertilizers inputs (r^2=1, a=0.33***). It means that each organic fertilizer application was followed by an additional release of mineral nitrogen equivalent to 33% of the mineral nitrogen contained in the fertilizers. To eventually refine the estimation of nitrogen inputs from the mineralization of organic fertilizers, we could collect site-specific parameters for poultry manure, instead of using generic data from literature. However, the good performances of STICS to simulate mineralization in a similar context (Sierra et al., 2010) and the satisfactory soundness of our simulations allowed us to conclude that no major uncertainty was associated with the estimation of this component of the nitrogen budget.

Harvested nitrogen

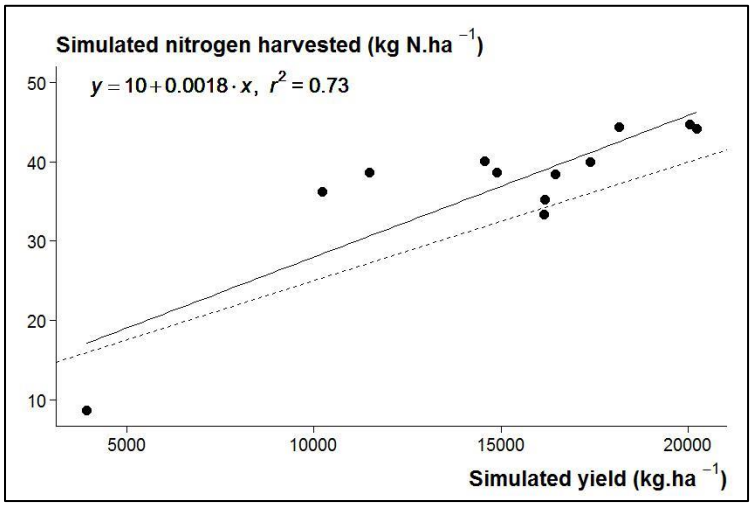


Figure 3.24 Simulated harvested nitrogen versus simulated yields

Harvested nitrogen constituted on average only 8% of the total output, and presented low variations across cropping systems (CV=26%). Quantities of nitrogen exported per hectare varied from 9 to 45, with an average of 37 kg $N.ha^{-1}$. These quantities were higher than average quantities (11 $kg.ha^{-1}$ on average, see chapter 2, Figure 2.5) calculated from measurements on urban gardens in

Niger (Diogo et al., 2010) and Burkina Faso (Lompo, 2012). Figure 3.24 shows exported nitrogen was correlated to simulated yields with an average nitrogen concentration of fruit estimated at 0.18%. This nitrogen concentration was similar to the 0.15% estimated from the literature (see chapter 2 and dotted line in Figure 3.24). We can extrapolate from the yields variations analysis (section 3.3.1) that variations of harvested nitrogen across cropping systems were multifactorial and very likely not representative for variations of observed data. To improve the simulation of harvested nitrogen we could collect data to calibrate cultivar-specific parameters regarding the production per plant and the nitrogen concentration of fruits. However to properly simulate tomato yields and subsequent nitrogen harvested for Beninese tomato cropping system, the model should also include losses due to parasitism stresses, as crop protection has often been shown inefficient in such context (Huat et al., 2013). To our knowledge, no model including parasitism stresses and associated crop protection is available.

Nr emissions

Nr emissions (NH_3, N_2O, NO_x and NO_3) constituted on average 21% of the total output and presented large variations across cropping systems (CV=122%). Ammonia emissions contributed significantly to the total emissions (58%) and quantities varied across cropping systems (CV=160%). The main influencing driver for the total NH_3 emission was the soil pH (Figure 3.15). We also found out that the total water input influenced NH_3 emissions from mineral fertilizers (Figure 3.16). However, as variations of NH_3 emissions from mineral fertilizers was much lower compared to variations of NH_3 emissions from organic fertilizers, we can conclude that water input resulted in low variations of the nitrogen budget when referring to its effect on NH_3 emissions from mineral fertilizers. A part of NH_3 emission from organic fertilizers variation remained unexplained using linear regressions. Considering the number of parameters involved to calculate the equilibrium between the different forms of nitrogen (Génermont & Cellier, 1997), influencing drivers were very likely multiple and thus difficult to identify. In STICS (Equation 6), NH_3 volatilized from organic fertilizers also depend on the volatilizable fraction of organic fertilizers, which was an estimated based on the water content of organic fertilizer. To eventually refine the estimation of NH_3 emissions we could collect site-specific parameters for poultry manure, instead of using generic data from literature. However, the likelihood between simulated NH_3 emissions and measurements from the literature (Predotova et al., 2010; Lompo et al., 2012) and the satisfactory soundness of our simulations allowed us to conclude than no major uncertainty was associated with the estimation of this component of the nitrogen budget.

Nr emissions from nitrification and denitrification processes contribute significantly to the total emissions (25%) but quantities did not vary much across cropping systems compared to other emissions (CV=75%). Nitrous oxide emissions presented contributed for 11%, however quantities varied across cropping systems (CV=130%). Nitrogen oxide emissions contributed for 14%, and quantities did not vary much across cropping systems (CV=60%).

At first sight, conditions seemed favorable for nitrification processes (high temperature, high water and nitrogen inputs). This assumption was validated as we obtained an average emission factor of 0.4% for NO_x (CV=7%), above the reference of 0.1% of the total N inputs, commonly used for NO_x (Nemecek and Kägi, 2007). We could identify soil pH as an influencing driver for nitrification. However, we could question the validity of our assumption to estimate NO_x emissions (0.05% of nitrified nitrogen) which was based on measurements in a very different context compared to our case study.

For N_2O emissions, the measurements from the literature in similar contexts (Predotova et al., 2010; Lompo et al., 2012), suggested that the emissions factor would be higher than the IPCC (IPCC, 2006) reference of 1% of the total N inputs for N_2O. However, we obtained an average emission factor of 0.3% for N_2O (CV=175%). A significant part of the difference between the IPCC reference and the simulated emission factors may results from the high contribution of soil organic nitrogen to the total nitrogen inputs. When only considering fertilizers as nitrogen inputs we obtained an average emission factor of 0.6% . We found out that for soils with moisture at maximal field capacity below 18%, the model STICS did not simulate any denitrification process. It can be explained by the effect of the field capacity in the calculation of the daily water-filled pore space (WFPS) as a function of soil water content (maximum corresponding to field capacity) and bulk density. Indeed, the denitrification, as an anaerobic process, does not occur until the WFPS exceeds 50–60%, then it increases exponentially as WFPS increases and levels off as soils approach saturation (Parton et al., 1998). In STICS, this threshold has been set to 62% based on the NOE equations (Hénault et al., 2005). Considering the low soil moisture at field capacity and associated bulk density of our soils most cropping systems cannot reach this threshold. This result was in favor of a satisfactory soundness of STICS results. However, on dryland cropping systems (Del Grosso et al., 2002) showed that a major source of model error (DNDC in this case) was the inability to correctly simulate WFPS. This can be explained by the difficulty in estimating amounts of water from the macroporosity. To improve our confidence on N_2O emissions we could collect data to validate the estimation of WFPS for our soils. To refine the estimation of N_2O emissions we could also provide site-specific data for $VPOTDENIT_P$. Finally Chen et al. (2008) underlined in a review of simulation approaches for N_2O emissions that an important challenge referred to the partitioning of gaseous N losses into N_2O and N_2. In STICS this ratio was fixed for denitrification ($RATIODENIT_P$) based on the literature (Garrido et al., 2002). However it was found that this ratio varied (Del Grosso et al., 2000) especially in irrigated vegetable cropping systems (He et al., 2009). This analysis suggested that important uncertainties were associated with the estimation N_2O emissions from denitrification. We can conclude that emissions from nitrification and denitrification cannot be validated at this stage.

Nitrate emissions contributed significantly to the total emissions (21%) and quantities varied across cropping systems (CV=179%). An important part of NO_3 emission variations could be explained by nitrification rates. To eventually improve our confidence on this variable we could collect data to

validate soil NO_3 concentration at the beginning and at the end of the crop. However, the performances of STICS to simulate nitrate leaching in contrasted contexts (Schnebelen et al., 2004; Jégo et al., 2008) and the satisfactory soundness of our simulations allowed us to conclude that no major uncertainty was associated with the estimation of this component of the nitrogen budget.

Qualitative assessment of the overall approach

Table 3.5 summarizes the performances of the suggested estimation methods for Nr emissions combining a crop model and a nutrient budget. The budget approach allowed us to identify the main fluxes of nitrogen, while the modeling approach allowed us to identify influencing driver for each important nitrogen flux and associated uncertainties. As a result the nitrogen budget of out-of-season tomato production in Beninese urban gardens was mainly influenced by 4 drivers:

1. The quantity of organic fertilizers because of its contribution to the total inputs and its effect on mineralized and volatilized nitrogen,
2. The volumes of water applied because of its effects on volatilized and leached nitrogen,
3. The soil pH because of its effect on volatilized and nitrified nitrogen,
4. The soil moisture at maximal field capacity because of its effect on denitrified nitrogen,

Regarding uncertainties, we can express reservations on the reliability of harvested nitrogen, N_2O and NO_x emissions. Regarding harvested nitrogen, a first lead for improvement consists in collecting cultivar-specific parameters. However the fundamental uncertainty associated to yield losses due to pest pressure cannot be undertaken with the current tool. Regarding N_2O, a first lead for improvement consists in collecting specific parameters for the potential of denitrification. However, considering the lack of references for N_2O and NO_x emissions in our context, further data to validate water and nitrogen dynamics would be needed to improve the quality of model outputs. For other emissions and fluxes, although we still have room for improvement in reducing operational uncertainties, the reliability of estimates could be considered as satisfactory. Back to the Life-Cycle Assessment framework, there was a last source of uncertainty which refers to the budget approach: what is the fate of the nitrogen surplus? To resolve this issue we could run multiple simulations using the nitrogen surplus as initial conditions until we reach a steady state. However this procedure would require additional data on agricultural practices for several full climatic years. This question regarding the limit of the system will be further discussed in chapter 5.

Table 3.5 Performances of the structured estimation methods for Nr emissions

	Influencing drivers	**Operational uncertainty**	**Fundamental uncertainty**
N mineralization	Organic fertilizer inputs	Site-specific parameters for poultry manure	
N harvest	Crop density Temperature and/or radiations Volumes of water inputs	Cultivar-specific parameters	Yield losses due to biotic stresses
NH_3 emissions	pH Volumes of water inputs	Site-specific parameters for poultry manure	
NO_3 emissions	Rate of nitrified nitrogen → organic fertilizers	Validation of soil nitrogen dynamic	
N_2O & NO_x emissions	Soil moisture at maximal field capacity Soil pH	Validation of on soil water dynamic Field-specific value for VPOTDENITP	$RATIODENIT_P$ $RATIONIT_P$ NO_x=0.05% of nitrified nitrogen
Combined approach	1. Organic fertilizer inputs 2. Volumes of water inputs 3. Soil pH 4. Soil moisture at maximal field capacity	1. Cultivar parameters 2. VPOTDENITP 3. Site specific parameters for poultry manure	1. Yield losses due to biotic stresses 2. Validation of soil water content dynamic 3. Validation of soil nitrogen content dynamic 4. Exploration of the fate for Δ SoilN?

3.4. Conclusion

In this paper we proposed a structured method for identifying main drivers for Nr emissions based on the association of a nitrogen budget and a dynamic crop model. The advantage of the crop model approach was to include a diversity of parameters influencing Nr emissions, according to up-to-date knowledge on biological processes involved. The budget approach allowed us to rank influencing drivers and uncertainties associated to the estimation of the different components of the budget according to their contribution to input and output fluxes. We underlined one additional uncertainty related to the fate of the nitrogen surplus.

We compared simulated yield to observed yield and underlined the error involved by the absence of biotic stresses within the crop model. Regarding emissions, we assessed the model soundness with regard to measured literature data and variation of influencing parameters. We found out that despite the many parameters identified for each process there were only a few drivers affecting heavily the components of the nitrogen budget across the 12 cropping systems. We calculated the nitrogen budget for the twelve cropping systems and qualitatively assessed the uncertainty linked to the estimation of major contributing components. For those components we provided recommendations to refine or improve the quality of the estimation based on our expertise on the model and the literature. Finally we identified and ranked four influencing drivers and five origins of uncertainty associated to the calculation of the nitrogen budget.

The sensitivity of Nr emissions to soil type and agricultural practices suggested potential variations of the associated environmental impacts. In the next chapter we will use the simulated emissions with the crop model STICS in the modeling of 8 agronomic scenarios representative for the variability of the environmental impact of one hectare of out-of-season tomato.

3.5. Reference

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Contribution du chapitre 3 et perspectives dans le cadre de la thèse

Ce chapitre présente une approche basée sur un bilan d'azote et l'usage d'un modèle biophysique pour analyser la variabilité des émissions d'azote réactif au champ. Les résultats de cette expérimentation avec le modèle candidat STICS ne nous ont pas permis de conclure que ce modèle permettrait de raffiner les estimations des émissions dans le cadre de l'ACV appliquée à notre cas d'étude. Ce résultat en demi-teinte résulte notamment de l'incapacité du modèle à prendre en compte les pertes de rendements liées à la pression parasitaire, principale source de variabilité des rendements dans notre cas d'étude. D'autre part la grande quantité de données nécessaires pour paramétrer le modèle (culture) et valider les sorties (émissions) limite le potentiel de ce type de modèle pour un usage en ACV et plus particulièrement en contexte tropicale ou ces données sont rarement disponibles. Néanmoins, la robustesse de ce modèle et l'analyse fine des variations d'émissions réalisée dans ce chapitre nous a permis d'identifier 4 principaux paramètres influençant les émissions dans notre gamme de situations pédoclimatiques. Ainsi nous pourront propager ces variations aux résultats d'ACV sur la base de scenarios virtuels dans le chapitre 4 (Figure 3.25).

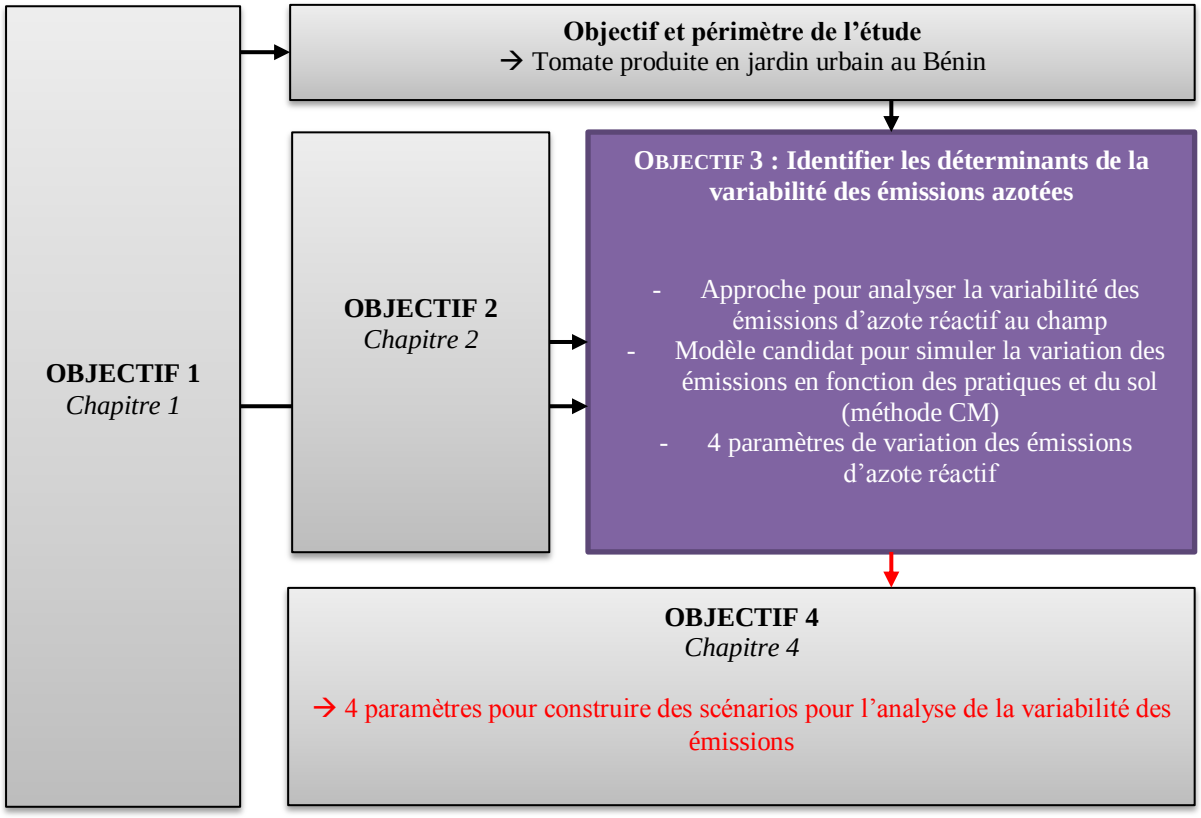


Figure 3.25 Résultats du chapitre 3 (boite violette) et contribution pour les chapitres suivants (rouge)

Chapitre 4. Exploration de la variabilité des impacts environnementaux calculés par l'analyse du cycle de vie de la tomate en maraîchage urbain au Bénin



Jardin urbain équipé d'un système d'irrigation par aspersion à Cotonou, Bénin

Avant-Propos

Nous avons montré dans les chapitres 2 et 3 que les systèmes de production de tomate en jardin urbain au Bénin étaient très diverses et avaient des flux très variables (intrants, rendements et émissions). Dans ce cadre, il est difficile d'en évaluer les performances environnementales uniquement sur la base d'un inventaire moyen. Nous proposons dans ce chapitre un grand nombre de résultats d'ACV pour plusieurs catégories d'impact, plusieurs scénarios et plusieurs choix méthodologiques. La figure 4.1 résume les données et les méthodes issues des chapitres précédents qui sont mises en œuvre dans ce chapitre. Au travers de cette diversité d'approches et de scénarios, ce chapitre vise à produire une connaissance robuste sur les impacts environnementaux de ces systèmes en prenant en compte la complexité mise en évidence précédemment.

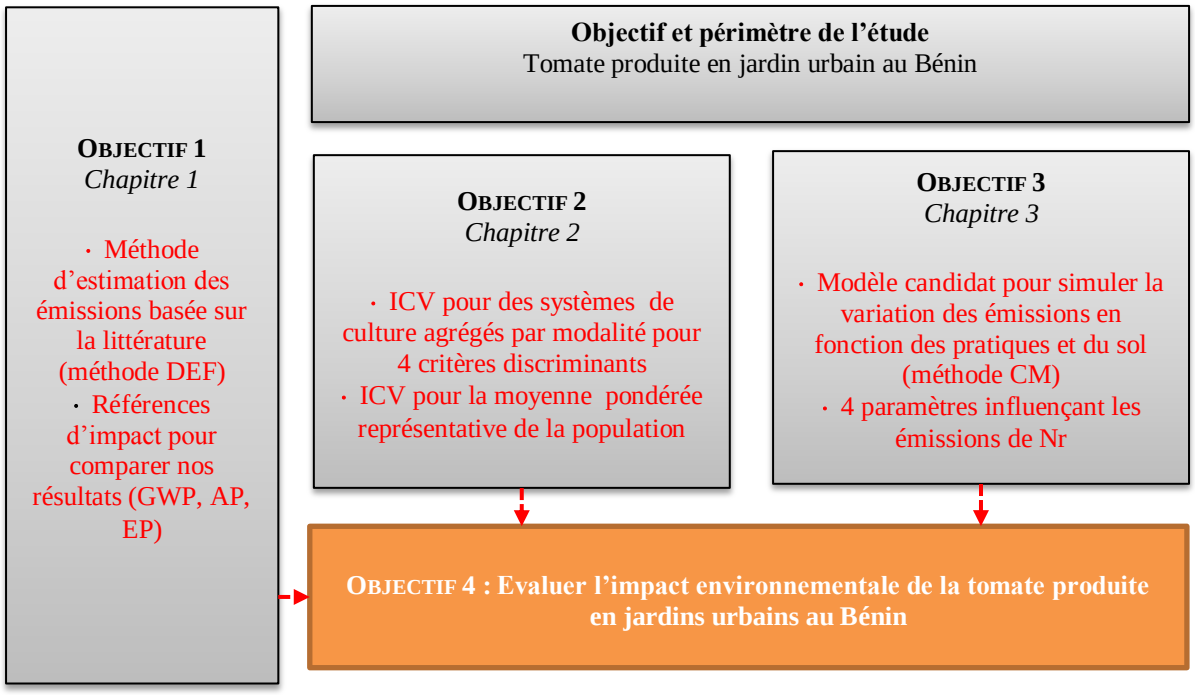


Figure 4.1 Les contributions des chapitres précédents (rouge) pour répondre à l'objectif 4

Résumé

A ce jour, aucune étude ne traite de l'évaluation environnementale des produits issus de l'agriculture urbaine Africaine, qui est en fort développement. L'application de la méthodologie Analyse du Cycle de Vie (ACV) à ces systèmes représente un défi scientifique par rapport à la prise en compte de la diversité des systèmes de production et de la variabilité des émissions au champ. Les objectifs de ce chapitre sont (1) de produire des références de résultats d'ACV pour la tomate en maraîchage urbain au Bénin et (2) d'explorer la variabilité des résultats associée à 4 facteurs influençant les émissions d'azote réactif au champ.

L'unité fonctionnelle choisie correspond à 1 kg de tomate. Les inventaires ont été construits à partir de données issues de 12 parcelles, regroupées selon 4 critères : le système d'irrigation, la source d'eau, la date de semis et la localisation géographique. Ces parcelles ont également été agrégées en une moyenne pondérée représentative de la fonction étudiée. Les catégories d'impact analysées sont la privation d'eau douce, le changement climatique, la toxicité humaine, la formation photochimique d'ozone, le potentiel d'acidification terrestre, le potentiel d'eutrophisation (marine et eau douce), et le potentiel d'écotoxicité (terrestre, marine et eau douce). L'analyse de sensibilité des émissions d'azote réactif a été réalisée à partir de 8 scénarios agronomiques (minimum et maximum pour 4 drivers) pour lesquels les émissions ont été estimées avec un modèle biophysique. Les drivers analysés sont le volume d'irrigation, la fertilisation azotée, le pH du sol et l'humidité maximal à capacité au champ.

Il apparait que les impacts environnementaux de ces systèmes de production de tomate en jardin urbain au Bénin sont supérieurs aux impacts de systèmes maraichers Européens. Les avantages du climat favorable au sud du Bénin et du faible besoin en infrastructure sont annulés par de faibles performances des systèmes d'irrigation, d'importantes émissions azotées et l'utilisation excessive d'insecticides. L'analyse de sensibilité montre que les résultats d'impact sont fortement sensibles aux variations d'émissions obtenues avec le modèle. L'analyse met notamment en évidence les risques de transferts de pollutions (ex : la réduction de la volatilisation induit une augmentation du lessivage) et la complexité des mécanismes biophysiques impliqués dans les émissions.

Il apparait que l'intégration des paramètres relatifs à l'irrigation et aux propriétés du sol (pH, capacité au champ) dans l'estimation des émissions au champ permettrait de produire des résultats d'ACV plus spécifiques et des recommandations plus fines pour réduire l'impact des systèmes étudiés.

Mots-clefs: ACV · Agriculture urbaine · Diversité des systèmes de production · Variabilité des émissions azotées

Investigation of the variability of life-cycle assessment of tomato production from Beninese urban garden

4.1. Introduction

At the beginning of the last century, only 14% of the world population lived in cities. Since 2007, the share of the world population living in cities reached 50% and is projected to reach 70% by 2030 (Kulikowski, 2007). This increase of population is correlated to an increase of the level of poverty in cities.

Urban agriculture in general, and urban gardening in particular host the most potential to increase citizen livelihood providing balanced food, reliable incomes and labor opportunities for urban citizens (Orsini et al., 2013). The rapid expansion of those production systems led to the intensification of agricultural practices to cope with the important land pressure on urban areas. Several studies underlined the environmental risk associated to this intensification leading to excessive use of fertilizers and pesticides (De Bon et al., 2010; Orsini et al., 2013). However, to date, a comprehensive environmental assessment, representative for a diversity of agricultural practices for urban horticulture in Africa is still lacking.

Life Cycle Assessment (LCA) is a normalized methodology (ISO 14040 and 14044) for the environmental assessment of products and services consisting of four steps: the goal and scope definition, the inventory, the environmental assessment and the interpretation. The relevance of LCA to study the environmental impacts of agricultural systems has been validated for a large number of products. In Europe, the use of detailed databases to characterize production system diversity (Mouron et al., 2006; Boulard et al., 2011) or the use of modeling tools to accurately estimate emissions (Bessou et al., 2013; Dufossé et al., 2013) led to significant improvement of the quality of LCA studies applied to agricultural systems.

By contrast, the application of LCA to vegetables in tropical contexts is almost inexistent. The achievement of life-cycle impact assessment results (LCIA) for vegetable production from urban gardens in Benin represents a vast scientific challenge. The lack of technical and environmental data and the absence of specific methods to estimate field emissions for tropical vegetable cropping systems reduce the reliability and the comparability of LCIA results for their products (chapter 1). The comparison of contrasted production systems is limited by uneven practices reasoning and agronomical performances (chapter 2). The data scarcity limits the possibility to accurately estimate N_r emissions with modeling tools (chapter 3). Finally, the lack of LCA studies applied to similar systems hinders the ability to conclude on the meaning of potential impacts. There is an urgent need for

transparent LCA studies addressing diversity and variability issues specific to vegetable systems from tropical contexts.

Based on a case study for out-of-season tomato in Benin showing extreme data scarcity, the objectives of this paper were (1) to produce novel LCIA results for vegetable production from urban gardens in Africa, and (2) to investigate the potential sources of LCIA variability associated to the estimation of N_r emissions.

4.2. Material and method

4.2.1. Goal and scope definition

Here we present the methodology of the study which corresponds to the first step of the LCA methodology as described initially in ISO 14044 (2006) and more recently in the ILCD Handbook (European Commission, 2010).

Goals of the study

The aims of this LCA study were to determine the environmental performances of out-of-season tomato production in Benin. To investigate the variability arising from both soil/climate drivers and management practices across cultivation fields associated to LCIA results, two different aspects were considered:

- The variability of the environmental impacts of cropping systems according to 4 key criteria: irrigation system, water resource, planting date and location,
- The sensitivity of results to 4 drivers affecting Nr emissions: water inputs, nitrogen inputs, soil pH and field capacity at saturation.

Functional unit

One kilogram of fresh yield and one hectare of tomato production were chosen as functional units as a reference to express input, output flows and subsequent LCIA results. Out-of-season tomato production in Benin shows a multiplicity of functions, i.e. producing fresh food all-year round, utilizing in the most effective manner one hectare of agricultural urban land, providing cash flow to farmers and employment for urban citizens. However, given the great variability of yield (chapter 2) and profit for those systems, choosing 1 kg of tomato or one unit of profit as unique functional unit would have introduced a bias in our LCIA results. Thus, we identified the land use as an interesting complementary function allowing to account for each individual farm whatever its yield and profit.

Data origin and quality

As presented in chapter 2, a typology-based procedure allowed exploring the actual diversity of cropping systems for out-of-season tomato production in Benin. Ten types were defined as the existing combination of modalities for the three selected technical criteria (irrigation system, water resource and planting date) within the population. Based on this typology, 12 cropping systems were selected and site-specific primary data were collected during the 2011-2012 dry season. The dataset obtained covered all cropping stages from sowing to harvest and included irrigation, fertilization and crop protection aspects. The quality of this dataset was assessed using the matrix of pedigree and scored (1, 3, 3, 1, 1) over the five criteria, namely reliability, completeness, technological, geographical and temporal correlation. From this dataset we built virtual systems in two steps (see chapter 2 for more details).

A first step consisted in building one synthetic and representative system called weighted mean. Our sample of 12 cropping systems allowed building 6 virtual scenarios on the basis of the typology of cropping systems recorded at population scale. Those 6 virtual scenarios resulted from the arithmetic average of data from monitored cropping system belonging to each of the 6 types for which cropping systems were sampled (1 virtual scenario = 1 type = 1 to 4 fields). The weighted mean was then calculated based on the weighted average of our 6 virtual scenarios according to their relative weight on an area-basis for the population (Table 4.1).

A second step consisted in building a range of systems representative for the diversity at population scale. Beyond the typology results as types and virtual scenarios, the 12 cropping systems may also be gathered by modality for the 3 technical criteria and their location representative of the diversity of cropping systems at population scale:

- The irrigation can be manual (14% of the population on area-basis) or done by hose requiring a pump (79% of the population),
- The water originated mainly from drilling (65%), but can also originate from depression (6%) or river (30%),
- The crop cycle can be set early (from September to October - 66%) or median (from November to December - 34%),
- The cropping systems can be located at Cotonou (19%), Ouidah (20%) or Grand-Popo (61%).
- This aggregation led to 10 modality-specific systems: 2 irrigation systems, 3 water resources, 2 planting dates and 3 locations.

Table 4.1 Weighting factors based on area distribution used to calculate the weighted mean and subsequent contribution of modalities within each criterion

Cropping systems	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Virtual scenarios		**S1**		**S3**	**S5**			**S6**			**S8**	**S10**
Weighting factors	2%	2%	2%	8%	45%	1%	1%	1%	1%	15%	15%	7%
Irrigation system	Manual (14%)				Hose (79%)							
Water resource	Depression (6%)			Drilling (65%)						River (30%)		Dr.
Planting date	Early (66%)					Median (34%)						E.
Location*	C	C	O	C	G	O	O	O	G	G	O	C

*C=Cotonou (19%), O=Ouidah (20%), G=Grand-Popo (61%)

Processes inventoried in the Ecoinvent database v2.2 were used as background data for energy production (Dones et al., 2007), material production (Hischier, 2007), fertilizer production (Nemecek & Kägi, 2007) and pesticide production (Sutter, 2010). Due to the complex logistic regarding the importation of energy material and inputs to Africa we did not include their transports in our inventories.

Methods for the estimation of field emissions

Gaseous emissions from energy combustion were calculated according to recommendations from Nemecek and Kägi (2007) for petrol. Given the lack of specific data on phosphorus and pesticide emissions in tropical conditions, we followed the recommendations from Nemecek and Kägi (2007) to estimate these emissions which corresponded to the more up-to-date guidelines for agriculture. We also based our estimation of NO_x emissions on these guidelines (21% of N_2O emissions). We used emission factors from IPCC (2006) to estimate direct (1% of nitrogen inputs) and indirect (1% of NH_3 emitted and 0.75% of NO_3 emitted) nitrous oxide emissions (N_2O) and to estimate nitrate (NO_3) leaching (as 30% of nitrogen inputs). Despite the lack of specificity of its emission factors, the IPCC report remains the most consensual method to estimate emissions in our context. For ammonia (NH_3) emissions from mineral fertilizers and poultry manure, emission factors from Bouwman and Van Der Hoek (1997) were used since they correspond to tropical conditions (4% for NPK, 20% for poultry manure and 25% for urea). Overall, to estimate fertilizer emissions we used the most recent or appropriate methods available, that we will further refer to as the default emission factor method (DEF).

Category of impact and LCIA methods

To assess the impact on resources, the water deprivation potential (WDP, expressed in m^3 equivalent) was calculated using the ReCiPe method (Goedkoop et al., 2009) associated to water stress index (WSI) from Pfister et al. (2011). The water deprivation potential results from the quantification of

water consumptions during the life-cycle, related to the water scarcity index (WSI) associated to the area where the water was withdrawn.

The following 9 impact categories selected as relevant to assess the environmental impact of studied systems were homogeneously calculated using the ReCiPe method.

To assess the impact on human health we selected three impact categories showing different contribution profiles. First, the climate change potential (CCP, expressed in kg CO_2 equivalent) was calculated using the equivalency factors from the 2007 IPCC report as recommended in the ReCiPe method. It results from the quantification of greenhouse gases emitted to air, related to their specific climate change potential over 100 years. Second, the human toxicity potential (HTP, expressed in kg 1,4-DB equivalents) results from the quantification of substances emitted to air, soil and water, relative to their specific human exposition route (exposure) and their toxicity (effect). Third, the photochemical ozone formation potential (OFP, expressed in kg NMVOCs[1] equivalent) results from the quantification of NO_x and NMVOCs emitted to urban air, related to their ability to increase the ozone concentration in the atmosphere.

To assess the impact on ecosystems we selected seven impact categories showing different contribution profiles. The terrestrial acidification potential (TAP, expressed in kg SO_2 equivalents) results from the quantification of substances emitted to the atmosphere which cause changes in levels of acidity when deposited on soils, related to their fate and effect on ecosystems. The freshwater eutrophication potential (FEP, expressed in kg P equivalent) and the marine eutrophication potential (MEP, expressed in kg N equivalent) result from the quantification of all emissions of P (FEP) and N (MEP) to air, water and soil and of organic matter to water, related to their fate in ecosystems and the sensitivity of receiving compartment. Terrestrial and freshwater ecotoxicity potentials (TETP and FETP expressed in kg 1,4-DB equivalents) result from the quantification of chemical substances emitted to soil (TETP) and water (FETP), related to their specific environmental persistence (fate), and toxicity (effect).

4.2.2. Inventory data

One inventory was built for each individual cropping system monitored as the assembly of:

- One hectare of nursery, weighted by the nursery area actually transplanted,
- One hectare of plot weighted by its duration,
- Irrigation infrastructure according to their duration of use.

[1] Non Methane Volatile Organic Compounds

The resulting 12 inventories were assembled into the weighted mean using the weighting factors presented in table 4.1. In parallel, the 12 inventories were assembled into 10 modality-specific inventories. In this section we present calculated inventory data in consistence with the LCIA results, that is aggregated according to the same rules. Table 4.2 presents the parameters used to build inventories. The nursery area used to transplant 100 square meters of tomatoes did not vary much across cropping systems. Main variations resulted from variations of plant density. Fuel consumption did not vary much except within modalities where cropping systems with pump were mixed with cropping systems without pump (manual irrigation due to one system with intermediary tank, early planting date and systems at Cotonou). Irrigation material was broken down into four elements, the pump, the pipe connecting the pump to the plot, the hose used to irrigate and the sprinklers. For each element, their specific weight and life-span were estimated based on farmer's declarations.

Table 4.2 Parameters used for the inventory modeling of production processes per modality of each criterion and for the weighted mean (WM)

		Irrigation System		**Water Resource**			**Planting Date**		**Location**			**WM**
		Manual	*Hose*	*Dep.*	*Drilling*	*River*	*Early*	*Median*	*Cotonou*	*Ouidah*	*GPP*	
Transplant on plots												
Nursery area ($m^2.100\ m^{-2}$)	*Mean*	0.005	0.009	0.005	0.009	0.005	0.008	0.007	0.007	0.004	0.01	0.007
	CV	47%	80%	57%	71%	110%	77%	73%	16%	105%	43%	71%
Nursery length (days)	*Mean*	43	23	36	29	28	38	23	45	23	25	27
	CV	33%	28%	15%	58%	15%	37%	30%	28%	33%	18%	48%
Plot density ($plant.m^{-2}$)	*Mean*	3.4	4.2	3.7	4.1	3.4	4.1	3.7	3.9	3.0	5.5	3.6
	CV	44%	41%	48%	43%	2%	48%	30%	27%	28%	37%	39%
Plot length (days)	*Mean*	76	74	75	76	68	77	71	73	69	85	78
	CV	10%	28%	12%	26%	23%	13%	29%	11%	32%	8%	20%
Irrigation energy												
Fuel consumption ($l.h^{-1}$)	*Mean*	0.3	1.5	0	1.4	1.3	0.8	1.3	0.5	1.1	1.7	0.9
	CV	200%	51%	-	55%	28%	140%	33%	115%	67%	69%	74%
Pump efficiency ($m^3.h^{-1}$)	*Mean*	450	2496	0	2067	3300	933	2579	900	1860	2724	1591
	CV	200%	30%	-	25%	13%	110%	31%	115%	67%	30%	50%
Irrigation material												
Pump (kg)	*Mean*	7	27	0	27	27	14	27	14	22	27	17
	CV	200%	0%	-	0%	0%	110%	0%	115%	56%	0%	35%
Pump Life-span (h)	*Mean*	828	5585	0	4093	7527	2416	4868	1152	3412	7346	3438
	CV	200%	58%	-	75%	35%	161%	59%	136%	104%	30%	75%
Hose (kg)	*Mean*	0	14	0	10	13	1	15	0	11	15	7
	CV	-	34%	-	79%	47%	245%	31%	-	70%	35%	60%
Hose Life-span (h)	*Mean*	0	4219	0	2677	5396	618	4304	0	3740	3612	2260
	CV	-	27%	-	72%	23%	245%	28%	-	62%	27%	51%
Pipe (kg)	*Mean*	9	136	0	106	162	32	146	29	87	172	81
	CV	200%	70%	-	89%	48%	119%	68%	129%	119%	48%	83%
Pipe Life-span (h)	*Mean*	552	5220	0	3947	5611	1210	5266	579	4990	3862	2873
	CV	200%	34%	-	59%	56%	168%	37%	188%	62%	24%	64%
Sprinkler (kg)	*Mean*	0	0	0	0.6	0	0.7	0	1.1	0	0	0.2
	CV	-	-	-	265%	-	245%	-	200%	-	-	278%
Sprinkler Life-span (h)	*Mean*	0	0	0	289	0	338	0	506	0	0	104
	CV	-	-	-	265%	-	245%	-	200%	-	-	278%

Table 4.3 presents a summary of inventory data for 10 modality-specific production processes and the weighted mean. Further details can be found in chapter 2 and appendix 4-5. Cropping systems with manual irrigation, early crop cycle or located at Cotonou presents particularly high insecticide treatment frequency index (TFI). Frequent insecticide treatments were mainly based on Cypermethrin (chapter 2, Figure 2.3). Fungicide and nematicide treatments were variable across cropping systems (CV = 126 and 169% respectively). While fungicide treatments appear unevenly distributed across cropping systems, the absence of nematicide treatments seems to be specific to cropping systems with manual irrigation and water from depression. While most of cropping systems mainly applied products based on terbufos, cropping systems with early crop cycles and located at Cotonou only applied products based on carbofuran (chapter 2, Figure 2.3). All cropping systems groups present organic and mineral fertilizer input. The location appears determinant regarding fertilization practices with high fertilizer use at Cotonou and low fertilizer use at Grand-Popo. This pattern suggests that land-pressure affects fertilizations practices (see chapter 2, section 2.3.3). Irrigation water volumes did not vary much across cropping systems (CV=77%). Cropping systems with manual irrigation and located at Cotonou high water use mainly due to one cropping system with intermediary tank used to stock pumped water before manual irrigation, which led to water losses through evaporation. By contrast cropping systems using water from river present low water use, mainly due to the soil type around river (clay and silt) which reduces water losses through drainage.

Table 4.3 Agricultural input data per modality of each criterion and for the weighted mean (WM).

		Irrigation system		**Water resource**			**Planting date**		**Location**			**WM**
		Manual	*Hose*	*Dep.*	*Drilling*	*River*	*Early*	*Median*	*Cotonou*	*Ouidah*	*GPP*	
***Crop protection** (TFI*)*												
Insect	*Mean*	12.0	5.7	7.6	7.4	7.8	9.9	5.1	11.6	4.9	6.5	8.2
	CV	85%	92%	82%	124%	47%	87%	107%	91%	99%	87%	85%
Disease	*Mean*	2.5	1.1	3.4	1.1	1.8	2.7	0.8	3.3	0.4	2.0	1.9
	CV	136%	131%	109%	134%	141%	99%	165%	95%	115%	90%	119%
Nematod	*Mean*	0.0	2.1	0.0	1.6	4.3	0.8	2.5	1.2	2.8	0.2	1.8
	CV	-	162%	-	172%	119%	245%	146%	200%	138%	173%	132%
***Organic fertilizers** (kg .ha^{-1})*												
N	*Mean*	433	259	419	342	226	384	301	543	385	3	221
	CV	104%	129%	131%	96%	141%	103%	115%	73%	83%	173%	157%
P_2O_5	*Mean*	151	91	147	120	79	134	105	190	134	1	105
	CV	104%	129%	131%	96%	141%	103%	115%	73%	83%	173%	115%
***Mineral fertilizers** (kg .ha-1)*												
N-NPK	*Mean*	64	55	77	79	36	82	61	112	62	31	52
	CV	76%	153%	65%	128%	141%	93%	150%	68%	159%	118%	140%
P_2O_5-NPK	*Mean*	30	24	37	43	16	47	27	64	29	14	30
	CV	63%	153%	47%	140%	141%	115%	150%	95%	147%	118%	125%
N-urea	*Mean*	56	37	61	15	101	38	43	56	0	87	37
	CV	111%	204%	125%	174%	141%	150%	187%	111%	-	120%	187%
Irrigation water ($m^3.ha^{-1}$)	*Mean*	6175	4791	4763	5992	2290	5472	4664	6671	4099	4546	4768
	CV	66%	73%	76%	62%	93%	64%	82%	48%	99%	74%	73%
Yields ($kg.ha^{-1}$)	*Mean*	6309	5123	5787	5219	3604	7733	2450	6307	1701	9121	9533
	CV	83%	144%	108%	147%	35%	105%	94%	83%	147%	115%	60%

*Treatment frequency index, see chapter 2, section 2.2.3 for more details

4.2.3. Design of and comparison with literature LCIA reference scenarios

For water deprivation potential (WDP), we can compare our LCIA results with literature scenarios based on water volumes weighted by the WSI (Water Stress Index) from Pfister et al. (2011) for tomato produced in Benin ("Benin x tomato") and tomato produced in Spain ("Spain x tomato"). We used yields from FAOSTAT for the year 2000 (FAO, 2013) to calculate water volume per hectare, consistently with data on water consumption used in the original paper (Table 4.4). It is important to notice that those literature scenarios only referred to water used for irrigation and not to water used in background processes. However, volumes of water used in the background processes within studied systems were low compared to volumes for irrigation water.

We can compare our LCIA results for global warming (GWP), acidification (AP) and eutrophication (EP) potentials with literature scenarios based on results reported in a review of LCA of vegetable products (chapter 1). The comparison with these average scenarios allowed us to get rid of the extreme variability of LCA results on a case-to-case basis (chapter 1, Figure 3.1) while highlighting major tendencies. The "Cold-greenhouse vegetable" scenario corresponds to the average impact from 12 cold greenhouse soil tomato systems extracted from 5 papers (Antón et al., 2005; Boulard et al., 2011; Martínez-Blanco et al., 2011; Romero-Gámez et al., 2012; Torrellas et al., 2012). The "Open-field tomato" scenario corresponds to the average impact of 46 open-field tomato systems extracted from 7 (Mila i Canals et al., 2008; de Backer et al., 2009; Martínez-Blanco et al., 2009; Martínez-Blanco et al., 2010; Martínez-Blanco et al., 2011; Romero-Gámez et al., 2012; Abeliotis et al., 2013). Main characteristics are summarized in table 4.4.

Table 4.4 Characteristics for literature vegetable scenarios and Beninese out-of-season tomato (this study)

Scenarios	**Benin x tomato**	**Spain x tomato**	**Cold-greenhouse vegetable**	**Open-field vegetable**	**Our study (WM)**
Yield (kg.ha^{-1})	5 399	60 469	119 808	41 582	9533
Irrigation water (m3.kg^{-1})	0.38	0.02	0.05	0.32	0.50
WSI	0.018	0.715	Ø	Ø	0.023
Energy consumption (MJ.kg^{-1})[1]	Ø	Ø	0.01	0.46	0.01
Mineral fertilizers (kg N.kg^{-1})	Ø	Ø	0.003	0.007	0.009
Organic fertilizers (kg N eq.ha^{-1})[2]	Ø	Ø	0.001	0.025	0.023

1. Quantities of energy were converted to megajoul based on specific energy density for fuel (36 MJ.l^{-1}), 2. Quantities of nitrogen equivalent for organic fertilizers were calculated using an average nitrogen content of 3%

4.2.4. Design and analysis of simulated cropping system scenarios with the crop model STICS

We used the crop model STICS (Brisson et al., 2002) as an alternative method to the DEF method to estimate N_r emissions for the scenario analysis. This method will be further called the crop model method (CM). This method aimed at including a diversity of specific parameters which influenced N_r emissions at the LCIA result level. Relevant impact categories for this scenario analysis were: Climate change Potential, Eutrophication potential and Acidification potential. STICS was selected for its generic aspect and its robustness to simulate crop yields (including open-field tomato), water and nitrogen dynamics for a range of crops and environmental contexts. Input data consists in four files: climate, soil, management and initialization. Climate data for Benin were provided by the local international research center named AfricaRice. Soil data were collected for the 12 cropping systems. Further detail on the model and input data can be found in the chapter 3 (sections 3.2.2 and 3.2.3) of this thesis.

Regarding simulated scenarios, we built first a reference scenario based on the environmental inventory of the weighted mean. To this end, we used the same aggregation procedure based on types and weighting factors to calculate soil parameters used as inputs for the crop model (see chapter 2, section 2.2.3). To build the daily management data used as inputs for STICS we considered:

- 55 irrigation events occurring every day from the transplanting date up to the harvest with same volume of water per event
- 2 applications of organic fertilizer occurring respectively 5 and 30 days after transplanting with the same nitrogen rate
- 2 applications of mineral fertilizer occurring respectively 10 and 25 days after transplanting with the same nitrogen rate.

Second, we selected 4 drivers responsible for important variations of N_r emissions (chapter 3, Table 3.5). The variation of nitrogen inputs affects all emissions through the variation of nitrogen quantities sensitive to emission processes and through the link between hexogen nitrogen and soil mineralization. The variation of water use affects a large range of variables: yield, volatilized nitrogen and to a lesser extent leached nitrogen. The variation of soil pH mainly affects the volatilization processes and the variation of soil moisture at maximal field capacity affects the denitrification process.

Finally, we built 8 scenarios based on minimum (Min) and maximum (Max) values regarding each driver independently (Table 4.5). Irrigation water use varied from 586 to 10,581 $m^3.ha^{-1}$ across the 12 cropping systems sampled. Both the number of irrigation events and the daily rates were correlated to this variable. We adapted the number of irrigation events to 26 for the minimum scenario and 77 for the maximum scenario. Nitrogen input rates varied from 21 to 1,184 kg $N.ha^{-1}$. The type of fertilizer

used, the number of fertilization events and the application rate were correlated to this variable. We used the overall distribution of fertilizer types shown by the weighted mean system (i.e. 71% as poultry manure, 17% as NPK and 12% as urea), and varied the number of fertilizer applications from 1 to 3 for organic products and from 1 to 4 for mineral products. Soil pH varied from 4.6 to 6.9. This parameter did not seem to be correlated to any other soil parameters, thus we could change it independently of the other soil properties. Soil moisture at maximal capacity (shortened as maximal soil moisture) varied from 9.6 to 61.6%. Unlike soil pH, this parameter was correlated to the clay content, the quantity of organic nitrogen and the field capacity at wilting point. We therefore used the actual values of maximal soil moisture for the two different soils of our sample.

Table 4.5 Key influencing drivers tested and corresponding favorable and unfavorable scenarios to emissions for the scenario analysis of LCIA results

Influencing drivers	**Influence on emissions**	**Scenarios**		**Correlated variables**
		Min	*Max*	
Water inputs ($m^3.ha^{-1}$)	The more water was supplied, the higher was the leaching rate	586	10 581	Irrigation events: 26 to 77 Daily rate
Nitrogen inputs (kg $N.ha^{-1}$)	The more nitrogen was supplied, the more N_r was emitted	21	1 184	Types of fertilizers used Fertilization events: 1 to 3 (organic) or 4 (mineral) Application rate
Soil pH	The higher was the soil pH, the higher was the volatilization rate	4.6	6.9	
Maximal soil moisture (%)	The higher was the maximal soil moisture, the higher was the denitrification rate	9.6	61.6	Clay content Soil organic nitrogen content Minimal soil moisture

Finally STICS did not simulate indirect N_2O emissions. We thus applied for those emissions the same emissions factors as for the DEF method (i.e. N_2O_i= 1% of NH_4 as simulated with STICS + 0.75% of NO_3 emissions as simulated with STICS).

4.3. Results

4.3.1. LCIA results across the modalities and the weighted mean

Figures 4.2 to 4.4 present the environmental impacts of the 10 modalities and the weighted mean calculated with the DEF method, expressed per kilogram of tomatoes. Each vertical panel presents one criterion to which modalities were associated. Each horizontal panel presents one impact category expressed in its reference substance. The figures also present the contribution analysis for each modality and for the weighted mean. We presented the contribution of irrigation practices through the energy production and use, the irrigation material production and the water consumption; the contribution of fertilization practices through the fertilizer production, N_r emissions and phosphorus emissions during the crop stage and emissions during the nursery stage; and the contribution of crop protection practices through the production and use of nematicides, fungicides and insecticides.

Categories mainly influenced by irrigation practices

Figure 4.2 presents LCIA results for five impact categories showing similar pattern of variation across modalities: water deprivation potential (WDP), climate change potential (CCP), human toxicity potential (HTP), photochemical ozone formation potential (POFP) and marine ecotoxicity potential (METP). Main contributors for these impact categories were associated to irrigation practices.

The weighted mean had a WDP of 0.014 m^3 eq. per kilogram of tomatoes. The major contributor was the water used for irrigation (71%). Despite low water volumes involved in the inventory of energy production and use and fertilizer production, their quite high contributions to WDP : 15% and 13%, respectively can be explained by the use of the World average WSI for background processes (WSI=0.669) which is higher than the local WSI used for irrigation water on farm (WSI=0.023 on average). The WDP varied from 0.009 (modality Grand-Popo) to 0.097 (modality Ouidah) m^3 eq. per kilogram across modalities.

The weighted mean showed a CCP of 3.08 kg CO_2 eq. per kilogram of tomatoes. The major contributor was the energy production and use for water pumping (84%). The second most important contributor was the nitrous oxide (N_2O) emissions (direct N_2O emissions with an 8% contribution and indirect N_2O emissions with 3%). The third contributor was the production of fertilizers (3%). The CCP varied from 1.33 (Depression) to 11.55 (Ouidah) kg CO_2 eq. per kilogram across modalities.

The weighted mean showed a HTP of 1,553.7 kg 1,4-DB eq. per kilogram of tomatoes. The only contributor was the energy production and use for water pumping. The HTP varied from 0.1 (Depression) to 4,465.3 (Ouidah) kg 1,4-DB eq. per kilogram across modalities.

The weighted mean showed a POFP of 0.006 kg NMVOC per kilogram of tomatoes. The major contributor was the energy production and use for water pumping (88%). The second most important

contributors were the nitrogen oxide (NO_x) emissions and fertilizer production with 4% each. The POFP varied from 0.002 (Depression) to 0.019 (Ouidah) kg NMVOC per kilogram across modalities. The weighted mean showed a METP of 0.16 kg 1,4-DB eq. per kilogram of tomatoes. The only contributor was the energy production and use for water pumping. The HTP varied from 0.01 (Depression) to 0.46 (Ouidah) kg 1,4-DB eq. per kilogram across modalities.

For the criteria: irrigation system, planting date and location, LCIA results variations resulted mainly from yield variations. Nevertheless, some discrepancies can be observed for WDP. The influence of irrigation volume led to similar WDP for the modalities of irrigation. In addition, there was also an effect of the water stress index associated to the location (WSI= 0.0129 at Grand-Popo vs. 0.0325 at Ouidah and Cotonou) which more clearly appeared when WDP was expressed per hectare of tomato production (Annex 8). Regarding the water resource criterion, cropping systems using water from depression presented the lowest CCP, HTP, POFP and METP thanks to the absence of energy consumption for irrigation. Despite higher yields cropping systems using water from drilling presented higher CCP, HTP, POFP and METP than cropping systems using water from river. These differences resulted from higher irrigation water volumes for cropping systems using water from drilling (5219 vs. 2290 $m^3.ha^{-1}$), because the clay and silt soils around rivers reduce water losses through drainage. These differences also resulted from lower fuel consumption, which can be attributed to lower energy needs to pump surface water than ground table water.

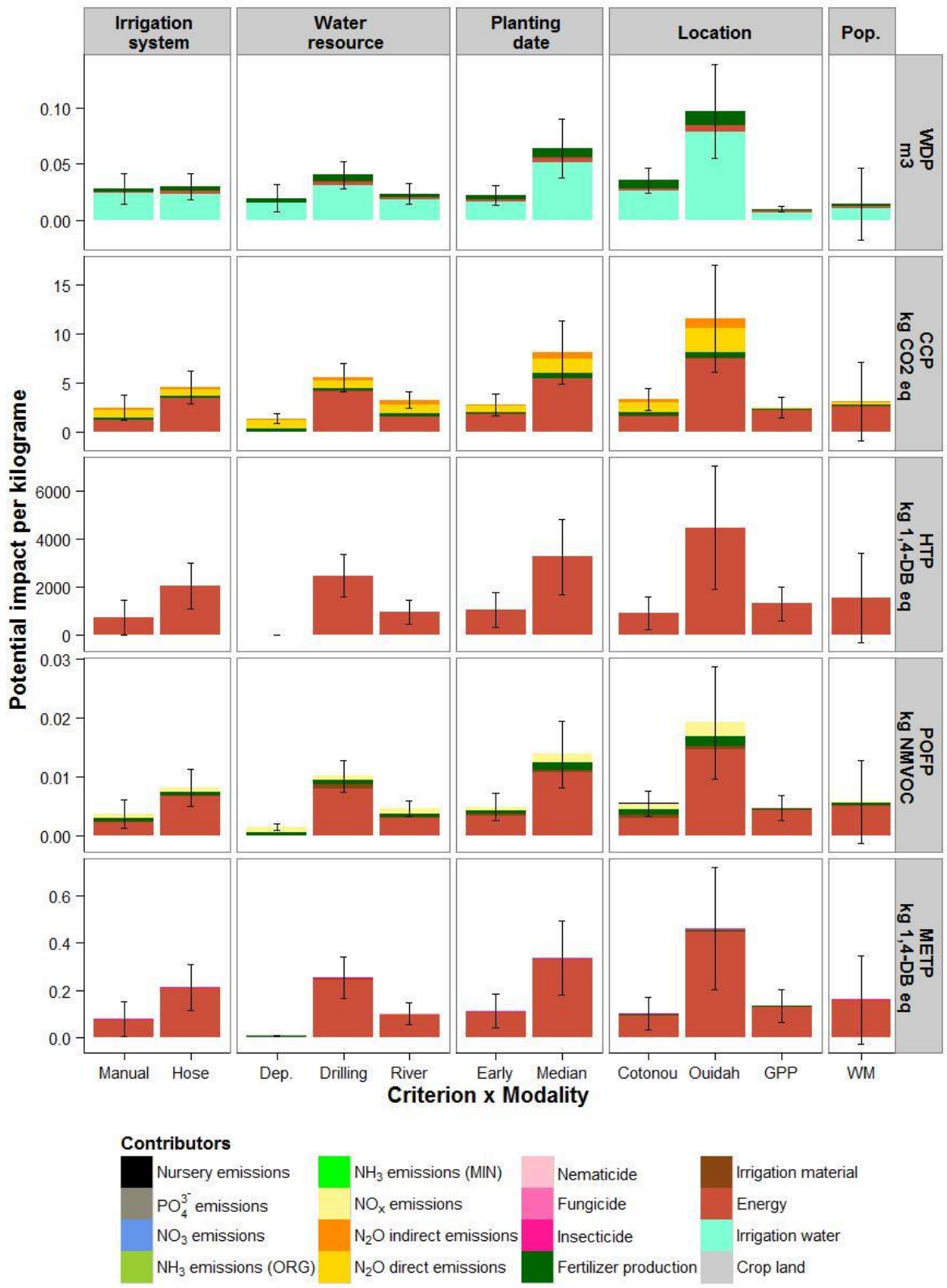


Figure 4.2 LCIA results from cradle-to-harvest per kilogram of out-of-season tomato in Benin of the 10 modalities accounting for 4 criteria, and the weighted mean for 5 impacts categories: water deprivation potential (WDP), climate change potential (CCP), human toxicity potential (HTP), photochemical ozone formation potential (POFP) and marine ecotoxicity potential (METP)

Impacts mainly influenced by fertilization practices

Figure 4.3 presents LCIA results for three impact categories showing similar pattern of variation across modalities: terrestrial acidification potential (TAP), freshwater eutrophication potential (FEP) and marine eutrophication potential (MEP). Main contributors for these impact categories were associated to fertilization practices.

The weighted mean showed a TAP of 0.02 kg SO_2 eq. per kilogram. The main contributor was the ammonia (NH_3) emitted from organic fertilizers (71%) and from mineral fertilizers applications (14%). Energy production and use and mineral fertilizers production also contributed to the TAP impact of the weighted mean (8% and 4% respectively). The TAP varied from 0.01 (Grand-Popo) to 0.16 (Ouidah) kg SO_2 eq. per kilogram across modalities.

The weighted mean showed a FEP of 0.0006 kg P eq. per kilogram. The main contributor was fertilizer production (71%). Phosphorus emission and energy production and use also contributed to the impact of the weighted mean (19% and 7% respectively). The FEP varied from 0.0004 (Grand-Popo) to 0.0033 (Ouidah) kg P eq. per kilogram across modalities.

The weighted mean showed a MEP of 0.009 kg N eq. per kilogram. The main contributor was nitrate emission (92%). The ammonia (NH_3) emitted from organic fertilizers also contributed to the impact of the weighted mean (5%). The MEP varied from 0.004 (Grand-Popo) to 0.086 (Ouidah) kg N eq. per kilogram across modalities.

For the planting date and location criteria, LCIA results variations resulted mainly from yield variations. Consequently systems located at Cotonou showed lower impact than systems located at Ouidah despite higher fertilizer uses. Regarding irrigation systems and water resources criteria there was no clear impact difference between modalities. However, when LCIA were expressed per hectare (Annex 8), we could identify the influence of mineral phosphorus use on FEP variations across modalities of water resources.

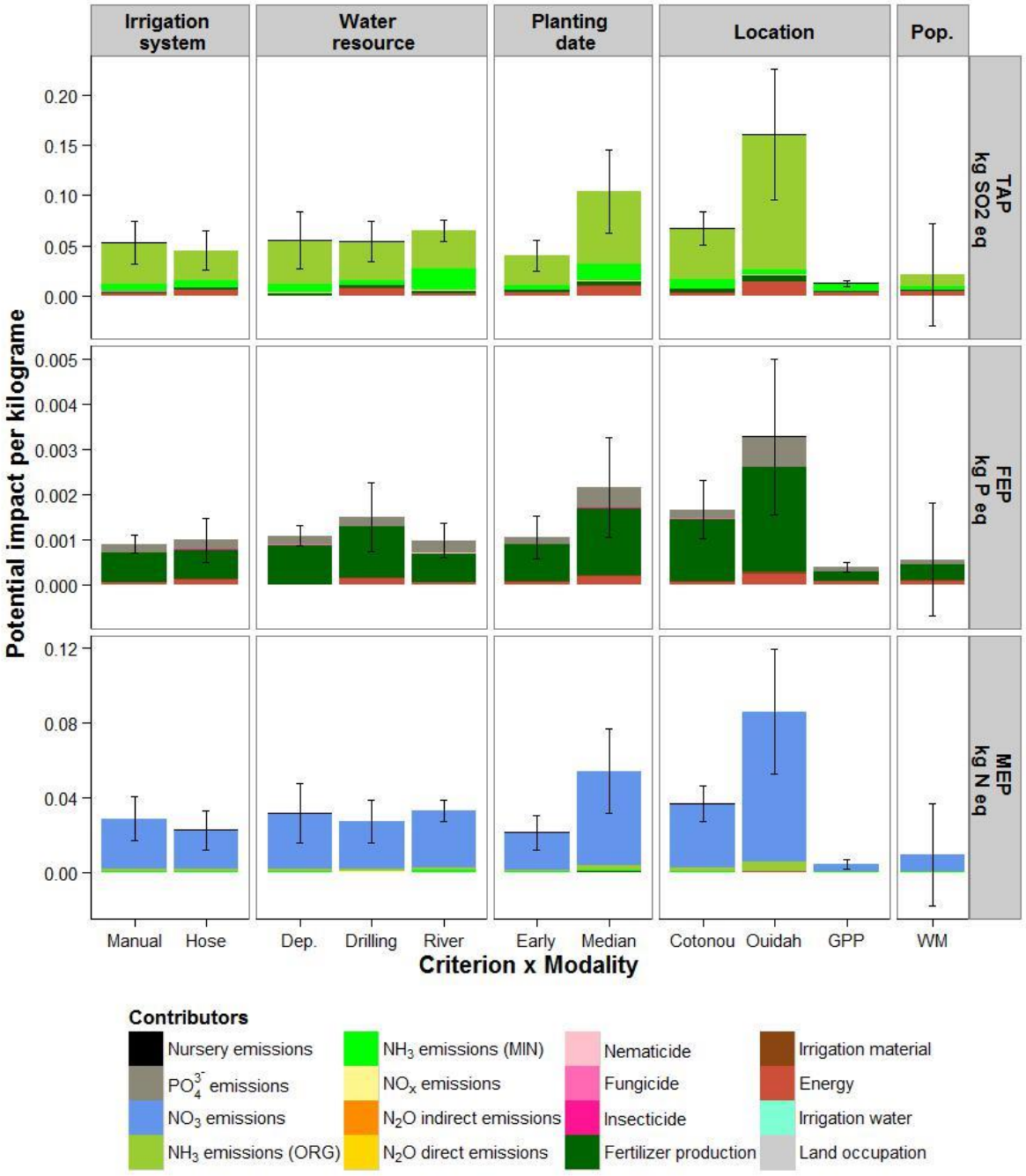


Figure 4.3 LCIA results from cradle-to-harvest per kilogram of out-of-season tomato in Benin of the 10 modalities accounting for 4 criteria, and the weighted mean; for 3 impacts categories: terrestrial acidification potential (TAP), freshwater and marine eutrophication potential (FEP and MEP)

Impacts mainly influenced by crop protection practices

Figure 4.4 presents LCIA results for two impact categories (TETP and FETP) each category showing a different pattern of variation across modalities.

The weighted mean showed a terrestrial ecotoxicity potential (TETP) of 0.21 kg 1,4-DB eq. per kilogram of tomato. The major contributor was the insecticide production and use (90%). Nematicide

and energy productions and uses contributed respectively for 7 and 3% of the total impact. The TETP varied from 0.16 (Grand-Popo) to 0.38 (Cotonou) kg 1,4-DB eq. per kilogram across modalities. Unlike other categories, TETP variations were not much driven by yield variations. Despite higher yield, cropping systems with manual irrigation systems showed higher TETP than systems with hose irrigation, due to greater use of insecticides. For the same reasons there was not much difference of TETP between modalities of water resource and planting date and cropping system at Cotonou presented higher TETP than cropping system at Ouidah. Despite lower yields, similar insecticide use and relatively high contribution of nematicides, cropping systems using water from river presented lower TETP. This can be explained by a lower share of cypermethrin which presents the highest toxicity potential over all active substances.

The weighted mean showed freshwater ecotoxicity potential (FETP) of 0.05 kg 1,4-DB eq. per kilogram of tomato. Similarly to the TETP, an important contributor was the insecticide production and use. However the contribution for the latter only reached 50%. This lower relative contribution of insecticide production and use resulted from the higher contribution of phosphorus emissions (35%), nematicide production and use (8%), and energy production and use (4%). The FETP varied from 0.04 (Grand-Popo) to 0.14 (Ouidah) kg 1,4-DB eq. per kilogram across modalities. FETP variations were very similar to TETP variations, i.e. mainly led by insecticide use in general and cypermthin use in particular. However, the use of nematicide in general and Terbufos in particular contributed to reduce the FETP difference between water resource modalities.

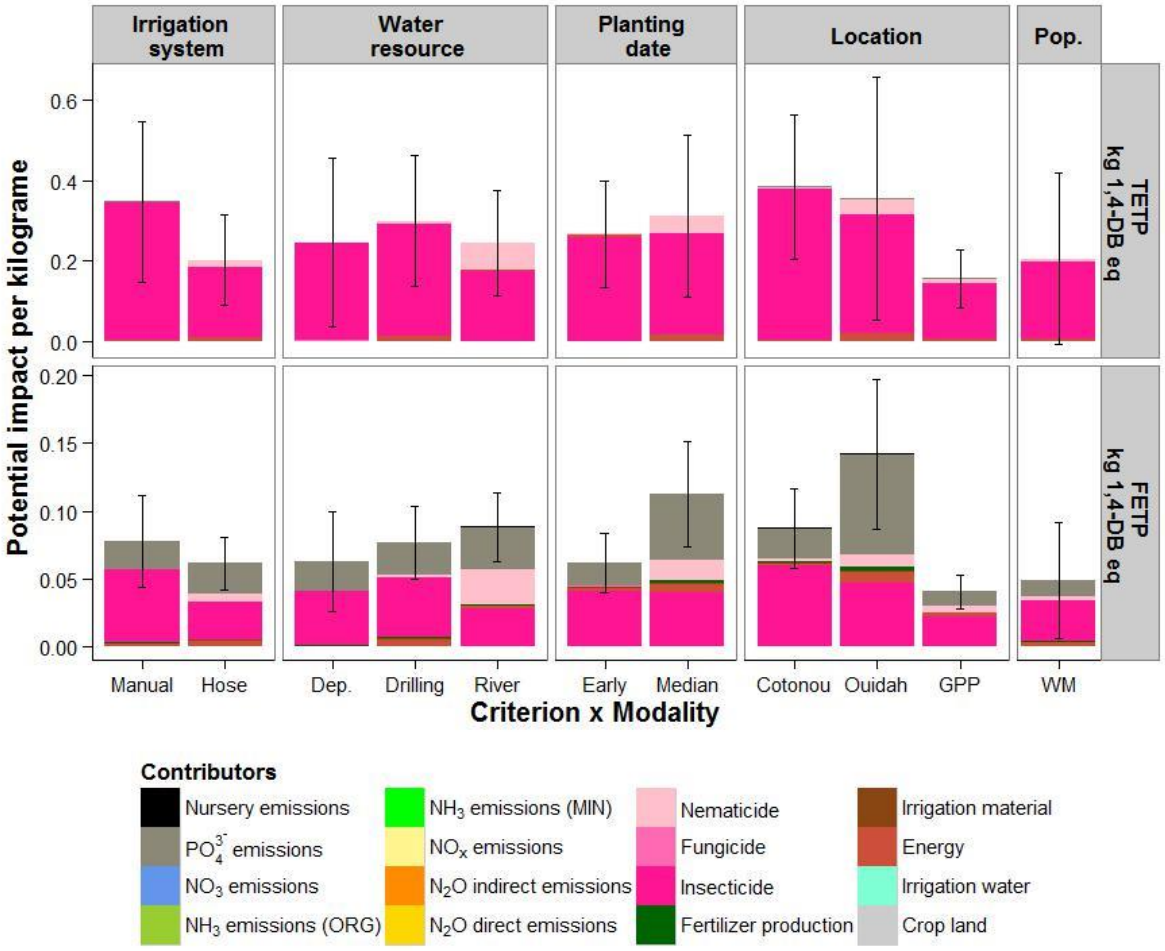


Figure 4.4 LCIA results from cradle-to-harvest per kilogram of out-of-season tomato in Benin of the 10 modalities accounting for 4 criteria, and the weighted mean; for 2 impacts categories: terrestrial and freshwater ecotoxicity potential (TETP and FETP)

4.3.2. Comparison with scenarios from the literature

Table 4.6 presents LCIA results expressed per kilogram of product for the 3 literature scenario and for the weighted meaning representative for the out-of-season tomato production in urban garden in Benin.

The WDP of one kilogram of out-of-season tomato in Benin (our study, weighted mean) was higher than the WDP of one kilogram of tomato in Benin ("Benin x tomato" scenario). This can be explained by the intensive irrigation requirement linked to the out-of-season production. Indeed, in Benin most of the tomato production is produced during the rainy season. Then, we also compared our results with a literature scenario accounting for tomato from Spain (Pfister et al., 2011) to assess the effect of changes in the value of WSI. The WDP of one hectare of out-of-season tomato in Benin (our study, weighted mean) was similar to the WDP of one hectare of tomato in Spain ("Spain x tomato" scenario). Indeed the higher water consumption for irrigation per kilogram of off-season tomatoes in Benin was compensated the low WSI associated to water extracted in Benin. Indeed, the coastal area of Benin presenting a sub-tropical climate, the impact of water use is lower than in drier areas such as Spain.

The GWP of one kilogram of out-of-season tomato in Benin was higher than the GWP of one kilogram of both European open-field vegetables ("Open-field x vegetable" scenario) and European greenhouse vegetables ("Cold-greenhouse vegetable" scenario). The yield difference is likely to be the main driver for such impact difference. However while the yield of European open-field vegetables was four times the yield of out-of-season tomato in Benin, the GWP increased by a factor 30. This difference can be explained first by higher irrigation water inputs for Benin tomato (0.50 for the WM vs. 0 0.32 $m^3.kg^{-1}$ for the "Open-field vegetable" scenario). Indeed, our systems being out-of-season they required more water than seasonal systems. The difference can also be explained by the source of energy used, which was an important driver for variations of the global warming potential across vegetable production systems (chapter 1). Unlike European producers Beninese producers had no access to electricity in their urban gardens. One unit of energy from fuel shows a higher GWP than one unit of the European electricity mix which often included electricity from nuclear power.

In agreement with the results for GWP, the AP for one kg of out-of-season tomato in Benin was higher than the AP for one kg of European vegetable whether under cold-greenhouse or open-field. As the AP increased by a factor 5 between the open-field vegetable scenario and our result, the yield difference is likely to be the main driver of such an impact difference.

In agreement with the results for GWP and AP, the EP for one kg of out-of-season tomato in Benin was higher than the EP for one kg of European vegetable whether under cold-greenhouse or open-field. The EP increased by a factor 15 between the "Open-field vegetable" scenario and our result despite the low contribution of energy production and use. This difference suggests that the high

impact of one kg of out-of-season tomato in Benin was due to high emission factors associated to volatilized and leached nitrogen emissions.

Table 4.6 LCIA results per kg of product for WDP, global warming, acidification and eutrophication potentials for literature vegetable scenarios and Beninese out-of-season tomato (this study)

Scenarios	Cold-greenhouse vegetable	Open-field vegetable	Benin x tomato	Spain x tomato	Our study (WM)
WDP (m^3 eq.)	Ø	Ø	0.007	0.015	0.014
Global warming (kg CO_2 eq.)*	0.37	0.11	Ø	Ø	3.08
Acidification (g SO_2 eq.)*	1.11	3.38	Ø	Ø	16.74
Eutrophication (g PO_4 eq.)*	0.37	0.58	Ø	Ø	8.15

***** For comparison purposes we calculated our LCIA results using the CML methodology.

Regarding ecotoxicity impact categories, there was few relevant LCIA result from the literature to compare our results with. Indeed, in the few papers dealing with ecotoxicity impacts of vegetable production systems (Anton et al., 2004; Boulard et al., 2011; Abeliotis et al., 2013) the methodology applied for LCIA calculation, and the estimation of emissions were not consistent with our case study. However, we can notice a very different contribution pattern between our result compared to Boulard et al. (2011) which presented a low contribution of pesticides to terrestrial ecotoxicity potentials (0.4% and 11% of the total impact for respectively the multi-span greenhouse and the tunnel systems). If we compare the terrestrial ecotoxicity potential for Beninese tomato to those presented by Boulard et al., (2011) for Tunnel system (excluding pesticide emissions), we end up with a higher impact for one kilogram of out-of-season tomato in Benin (0.0047 vs. 0.21 kg 1,4-DB eq. per kilogram for the weighted mean). This suggests that our results were particularly high which can be explained by the frequent use of pesticides, and especially of some insecticides with very high terrestrial ecotoxicity potential such as cypermethrin. We can notice the change in the scale of the y axis between terrestrial and freshwater ecotoxicities (Figure 4.4). It suggests a higher terrestrial ecotoxicity potential of crop protection practices compared with their freshwater ecotoxicity potential. The assumption of a 100% emission to soil used in this study might lead to an overestimation of the terrestrial ecotoxicity potential of some active substances. Indeed, simulations with recent estimation methods such as the Pest-LCI model (Dijkman et al., 2012) suggest that an important part of pesticides do not reach the soil, either because of crop interception or drift during spreading. Part of it is also biodegraded.

4.3.3. Scenario analysis to 4 drivers influencing N_r emissions

Variability of N_r emissions

Choices we made regarding the estimation of N_r emissions contributed largely to the high impact shown by one kilogram of out-of-season tomato in Benin regarding CCP, AP and EP. In the following section we will analyze the sensitivity of these three impact categories to N_r emissions within the

range of variations for the 12 surveyed fields. To consider the variations of N_r emissions, we selected 4 drivers to build 8 agronomical scenarios for which we simulated N_r emissions using the crop model STICS. Table 4.7 presents the N_r emission factors simulated with the crop model method for the weighted mean and the minimum and maximum scenarios for the 4 drivers compared with N_r emission factors obtained with the default emission factors (DEF).

Table 4.7 shows that using the crop model STICS to estimate the emissions for the weighted mean led to a decrease of the emission factors for N_2O (from 1 to 0.3%), for NH_3 from organic fertilizers (from 20% to 12%) and, for NH_3 from mineral fertilizers (from 11 to 3%). The volatilization rate was limited by the effect of soil pH, while the denitrification rate was limited by the effect of maximum soil moisture (chapter 3). By contrast, the leaching emission factor was increased (from 30% to 45%) due to large water applications on soils sensitive to drainage (if water is not stored in soil, it drains).

Table 4.7 N_r emission factors obtained with the Default Emission Factors (DEF) approach (calculated for the weighted mean only) and estimated with the crop model method (CM) for the 8 agronomic scenarios, minimum (Min) and maximum (Max)

	Weighted Mean		**Nitrogen rate**		**Water volume**		**Soil pH**		**Maximum soil capacity**	
	DEF	*CM*	*Min*	*Max*	*Min*	*Max*	*Min*	*Max*	*Min*	*Max*
N_2O	1.0%	0.3%	2.0%	0.2%	0.2%	0.3%	0.3%	0.3%	0.2%	3.7%
NH_3 from organic fertilizers	20%	12%	14%	12%	17%	12%	1%	51%	23%	4%
NH_3 from mineral fertilizers	11%	3%	4%	2%	3%	3%	0%	6%	3%	3%
NO_3	30%	45%	0%	53%	0%	64%	42%	29%	71%	0%
Simulated yields ($kg.ha^{-1}$)		17,824	17,794	17,848	14,693	18,149	17,812	17,824	17,870	17,706

Regarding agronomic scenarios (Table 4.7), the variation of nitrogen rate led to a variation of all emission factors. As expected, the NO_3 emission factor was lower for the minimum scenario and higher for the maximum scenario compared to the weighted mean (leaching rate depends on the soil nitrate concentration during drainage events). By contrast, emission factors for N_2O and NH_3 were higher for the minimum scenarios than for the weighted mean. Similarly, N_2O and NH_3 emission factors were lower for the maximum scenario than for the weighted mean, but not for NH_3 from organic fertilizers which remained constant. This means that changes in fluxes of N_2O and NH_3 emissions were not proportional to changes of N rate. This can be explained by the large contribution of mineralized nitrogen to emissions.

As expected the variation of water use led to a variation of NO_3 emission factors from 0 to 64% of the total nitrogen fertilizer input. The minimum scenario showed a higher NH_3 emission factor for organic fertilizer, however N_2O and NH_3 (min) emission factors were not much influenced by the irrigation volume.

Regarding the effect of soil pH as expected, the higher was the soil pH, the higher were the NH_3 emissions. NH_3 emission factors for organic fertilizers varied from 1 to 51% and from 0 to 6% for mineral fertilizers. However, we noticed an unexpected effect on the NO_3 emission factor which decreased compared to the weighted mean to reach 29% for the maximum scenario.
For soil moisture at maximum field capacity only the maximum scenario led to an important variation of the denitrification rate. Thus, N_2O emission factor reach 3.7% for the maximum. However, we noticed that the variation of maximal soil moisture led to an opposite variation of NO_3 emission factor (from 71 to 0%) and an opposite variation of NH_3 emission factor from organic fertilizer (from 23 to 4%). The increase of leaching rate when maximal soil moisture decreased resulted from the higher amount of drained water. The negative correlation of volatilization rate for organic fertilizers to maximal soil moisture was more unexpected and probably resulted from the complex mechanism involved in the acid-basic equilibrium governing the speciation of nitrogen in soil (chapter 3). Finally, we can notice that the water volume was the only driver responsible for important yield variation (from 14 693 to 18 149 $kg.ha^{-1}$ vs. 17 824 $kg.ha^{-1}$ for the weighted mean.

Variability of LCIA results
We then propagated this N_r emission variability into LCIA results for the 3 impact categories impacted by N_r emissions, namely climate change potential (CCP), terrestrial acidification potential (TAP) and marine eutrophication potential (MEP). Figure 4.5 presents the relative LCIA results for the maximum (Max) and minimum scenarios (Min) for each influencing driver independently. Results are expressed as a percent of the impact for the weighted mean calculated with the crop model method for N_r emission estimation. Similarly to figures 4.2 to 4.4, figure 4.5 also presents the contribution analysis for each scenario and each impact category.

For CCP, the most influencing driver was the nitrogen rate, because of higher indirect N_2O emissions resulting from the increase in NH_3 and NO_3. The CCP of the minimum scenario only reached 83% of the CCP of the weighted mean, while the maximum scenario presented a CCP 3 fold higher than the weighted mean. The second most influencing drivers was the water volume, due to the high contribution of energy consumptions for irrigation which was correlated to water inputs (Figure 4.2). The CCP of the minimum scenario only reached 43% of the CCP of the weighted mean, while the maximum scenario presented a CCP 2.6 fold higher than the weighted mean. The increase of CCP for the maximum scenario can be explained by both the increase of energy consumption and the increase of indirect N_2O emissions due to the increase of NO_3 emissions. As expected for the maximum soil moisture driver, the maximum scenario showed a higher CCP (+38%) due to higher N_2O emissions. However the minimum scenario showed an even higher difference to CCP than the favorable scenario (+62%) due to higher indirect emissions. For the soil pH driver, both the minimum and the maximum

scenarios showed higher CCP compared with the weighted mean (+34% and +26% respectively) due to indirect emissions either from NO_3 emissions (minimum scenario) or from NH_3 emissions (maximum scenario).

For TAP, the most influencing driver was the nitrogen rate, due to the high contribution of both NH_3 emissions and fertilizers production. The impact of the minimum scenario only reached 37% of the weighted mean, while the maximum scenario was 3 fold higher. As expected, the TAP was also largely influenced by the soil pH. The impact of the minimum scenario only reached 43% of the weighted mean, while the maximum scenario was 2.9 fold higher. To a lesser extent, AP was influenced by maximal soil moisture, with a higher TAP for minimum scenario (+49%) and a higher TAP for the maximum scenario (-35%) compared to the weighted mean. Both the minimum and the maximum scenario for water input exceeded the impact of the weighted mean (+62% and +30% respectively) due to higher NH_3 emissions.

For MEP, the most influencing driver was the nitrogen input, due to the high contribution of NO_3 emission. The MEP of the maximum scenario was 4.5 fold higher than the weighted mean, while the minimum scenario reached only 3% of the MEP of the weighted. By contrast for the maximum soil moisture driver, the MEP of the maximum scenario reached only 3% of the MEP of the weighted mean, while the MEP of the minimum scenario was 1.6 fold higher. Similar MEP results were found for the scenarios related to water volume and soil pH compared to the weighed mean. For the soil pH driver it can be explained by a compensative variation in NH_3 and NO_3 emissions associated to a relatively low contribution of NH_3 emissions. For the water volume driver, this absence of effect resulted from the variation of yields which softened the effect of the higher NO_3 emissions.

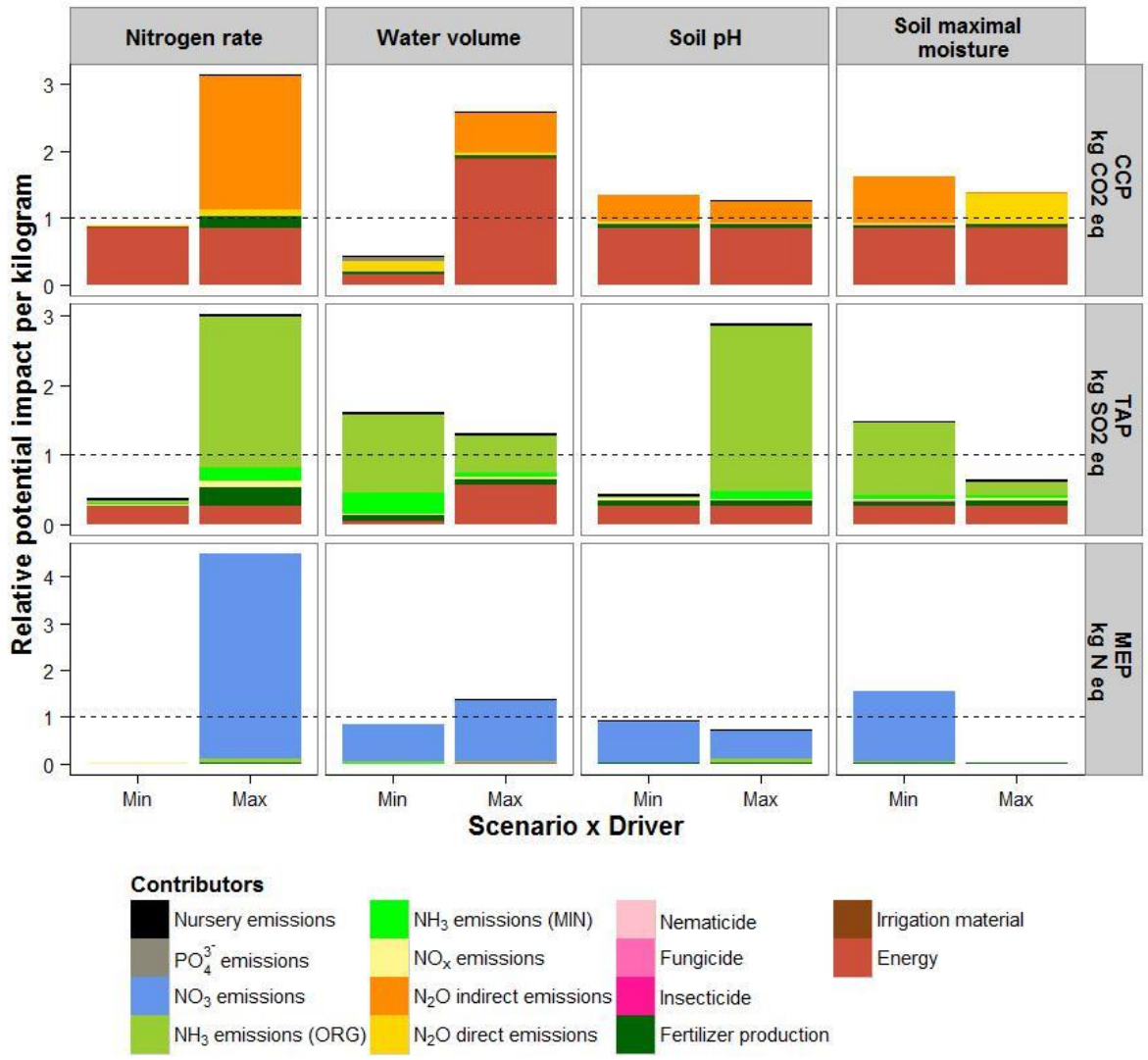


Figure 4.5 Scenario analysis for climate change potential (CCP), terrestrial acidification potential (TAP) and marine eutrophication potential (MEP) to minimum (Min) and maximum (Max) scenarios for 4 drivers influencing N_r emissions: nitrogen rate, water volume, soil pH and maximum soil moisture

4.4. Discussion

4.4.1. Impact of the irrigation management

In Benin, urban gardeners grow tomato out-of-season i.e. from September to April while tomato is often grown during the season from April to August in other urban gardens in Africa (Diogo et al., 2010; Sangare et al., 2012). The importance of out-of-season tomato production in urban gardens in Benin results from the high competition with extensive seasonal production in rural area with no irrigation system and low fertilization management, which reduces the profitability of tomato for urban gardeners during the season. As an out-of season production, the advantage of the studied system was its localization under a sub-tropical climate. Despite lower yields and higher water volume, out-of-season tomato showed similar water deprivation potential compared to the Spanish scenario due to the low WSI (Water Stress Index) calculated for this area (Pfister et al., 2011). However, we could question the validity of an annual WSI, especially for a short-cycle crop in a region where the water availability can vary from scarce to excessive within a climatic year. Monthly WSI would allow a better capturing of water scarcity issues along the year (Pfister & Baumann, 2012) and especially when the out-of-season tomato crop is grown.

Being under a sub-tropical climate prevents from the need for any specific infrastructure, which potentially reduces climate change potential compared to average European out-of-season vegetables grown under greenhouse. However, its low yield and high fuel consumption (high water inputs and low equipment efficiency) led to higher climate change potential than the average European vegetable whether under cold greenhouse and open-field. More generally, the major effect of irrigation management on CCP was confirmed by the scenario analysis with STICS. It also demonstrated that parameters referring to irrigation practices should be systematically included for the estimation of N_r emissions. Such inclusion may allow optimizing the irrigation management to reduce the water deprivation potential, the climate change potential, as well as the acidification and eutrophication potentials.

4.4.2. Impact of fertilization management

Similarly to other urban gardens in West Africa (Diogo et al., 2010; Sangare et al., 2012), Beninese urban gardens showed intensive fertilization practices and relatively low yields. Organic and mineral fertilizers inputs provide nutrient amounts largely above crop exportations (chapter 2). As a result, average nutrients inputs per kilogram of tomato for our systems were higher than average nutrients inputs for European open-field vegetables. Acidification and eutrophication potentials were higher than that for European vegetable scenarios. This can be explained by the high emission factors associated to N_r emissions which were selected to better represent our tropical context (Bouwman & Van Der Hoek, 1997). On this respect, the emission factors we selected from Bouwman and Van Der Hoek (1997) were above the emission factors calculated with STICS for the weighted mean. It

suggests that in most situations of our sample, the emission factor associated to volatilization would be lower than the reference from the literature and that it would be desirable to include key parameters associated to the volatilization process in the estimation method. The scenario analysis using STICS for calculating N_r emissions showed that the main changes in acidification and eutrophication potentials actually resulted from the variability of nitrogen input rates. The last point underlined by the scenario analysis with STICS was the compensation between emission processes. Our results suggested that reducing the volatilization rate may increase the leaching rate, due to higher soil nitrogen content (compensative losses). Similarly, reducing the denitrification rate by reducing water logging (low soil moisture capacity) led to higher leaching rate by increasing drainage and also led to higher volatilization rate (this aspect remains unexplained though). These opposite behaviors for emissions highlighted the complexity of bio-physical mechanisms involved in the processes of N_r emissions. More generally these results demonstrated that environment-friendly fertilization practices could only be achieved by considering the irrigation management and soil properties.

4.4.3. Impact of crop protection management

Similarly to other urban gardens in West Africa (Cissé et al., 2003; Ahouangninou et al., 2011), Beninese urban gardens showed intensive crop protection practices in terms of application frequency and applied rates. Our results showed that studied system had particularly high terrestrial and freshwater ecotoxicity potentials compared to European cold-greenhouse production systems (Boulard et al., 2011). This difference can be partly explained by the emission estimation method we chose which considered 100% of pesticides were emitted to soil (Nemecek & Kägi, 2007). Recent developments showed that a significant part of pesticides applied can be absorbed and biodegraded on plant leafs or in soils (Dijkman et al., 2012). However, we could also explain these large impacts by the use of certain active substances with high terrestrial and freshwater ecotoxicity potentials such as the terbufos and the cypermethrin. Our results suggested that the terbufos was even more critical than the carbofuran (also a nematicide) often quoted as the most dangerous product for the environment (Ahouangninou et al., 2012). The frequent use of cypermethrin at high rates was of particular concern regarding its persistence in aquatic ecosystems and its high toxicity (Feo et al., 2010). Finally, in our study, we did not consider the effects of pesticide residues on human health (Margni et al., 2002). Considering the important pesticide residues found on vegetables from Beninese garden by Ahouangninou et al. (2012), this aspect appears as an important limitation of our results and would warrant further research.

4.5. Conclusions

We assessed the impact of one kilogram of out-of-season tomato production in Benin from cradle-to-harvest. To this aim we calculated life-cycle impact assessment (LCIA) results for 10 modalities and the weighted mean with default emission factors for field emissions. The 10 modalities presented different characterization profiles for the 10 impact categories considered. The analysis of the representative weighted mean and the diverse modalities allowed identifying hot-spots for one kilogram of out-of-season tomato production in Benin. The results for our weighted mean were compared with reference results for vegetables from the literature, highlighting the high environmental impact associated to tomato production in urban gardens in Benin. Then we analyzed the sensitivity of 3 impact category results (CCP, AP and EP) to 4 drivers influencing N_r emissions. This analysis allowed us to underline the variations associated to LCIA results in order to refine the interpretation of these results and suggest improvements for emission estimation methods.

We can conclude that the LCA results of out-of-season tomato production in Benin were diverse and highly variable. We highlighted the fuel consumption, organic fertilizer use and insecticides applications as major hot-spots for the studied function. Some variations due to management practices and local drivers could be identified and quantified. We highlighted the major effect of irrigation on all N_r emissions, which stresses the need of including systematically this parameter into the emission estimation methods dedicated to irrigated systems. We also demonstrated that soil parameters played a significant role for the estimation of N_r emissions. Finally, our results also revealed weaknesses within the approach to estimate emissions based on emission factors from nitrogen fertilizers as it does not involve nitrogen inputs from soil mineralization which was a major input for our systems.

Based on the agronomical analysis of systems (chapter 3) and our LCIA results (this chapter) we propose to explore in the general discussion potential margins for improvement for producers to reduce the impact of their systems. We underlined methodological limitations for the application of LCA to tomato systems grown on urban gardens in Africa. Based on our expertise on the inventory modeling (chapter 3), on the analysis of N_r emissions (chapter 4) and on LCIA results, we will formulate in the general discussion scientific prospects which could allow improving the robustness of LCA applied to such systems.

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Chapitre 5. Discussion générale



Irrigation au Tuyau sur une parcelle de tomate en jardin urbain à Ouidah, Bénin

5.1. Introduction

Le taux de croissance urbain a atteint 4.6% en Afrique sub-saharienne, soit trois fois le taux de croissance urbain des pays développés (UN-HABITAT, 2013). Ce changement dans la répartition de la démographie mondiale s'est accompagné d'une augmentation de la pauvreté urbaine.

L'agriculture urbaine en général et l'horticulture urbaine en particulier est l'une des stratégies largement développées, notamment en Afrique, pour réduire la pauvreté et améliorer le bien-être des habitants des villes (Orsini et al., 2013). En effet les fruits et les légumes représentent une composante importante pour un régime alimentaire équilibré (minéraux, vitamines et autres éléments antioxydants). En plus de leur valeur nutritive, les fruits et légumes sont des cultures à haute valeur ajoutée qui peuvent constituer une source de revenus pour les petits agriculteurs, et particulièrement pour les femmes.

Aujourd'hui, la promotion de l'agriculture urbaine est freinée par le manque de données complètes et fiables sur ces systèmes de production (FAO, 2012; Orsini et al., 2013; World Bank, 2013). En effet, ce constat sur l'importance de l'agriculture urbaine renouvelle les enjeux associés à l'usage (souvent non maitrisé) des fertilisants minéraux et organiques et des produits phytosanitaires. Plusieurs études mettent en évidence les risques potentiels de l'agriculture urbaine sur la santé humaine et sur l'environnement (De Bon et al., 2010; Drechsel & Dongus, 2010; Ahouangninou et al., 2012). Néanmoins, la mise en œuvre de l'évaluation environnementale de ces systèmes, notamment par l'analyse du cycle de vie, soulève de grands défis scientifiques parmi lesquels la prise en compte de la variabilité de ces systèmes est sans doute la plus importante. La question centrale dans cette thèse a ainsi été : comment prendre en compte la variabilité du sol, du climat et des pratiques pour la réalisation d'ACV fiables ?

Pour ce travail, nous avons adopté une démarche intégrative combinant l'analyse agronomique (typologie, modélisation, indicateurs simples) et l'analyse de cycle de vie (objectifs et cadre de l'étude, inventaires et évaluation de l'impact). Nous avons travaillé à l'échelle de la parcelle et du cycle cultural. Nous étions conscients dès le départ des fortes interactions à l'échelle supérieure, à savoir l'exploitation que ce soit spatialement avec les autres activités de l'exploitation agricole ou temporellement avec les autres composantes de la rotation. Néanmoins, la parcelle est la plus petite unité de raisonnement de l'itinéraire technique, ainsi que l'échelle de simulation avec le modèle biophysique. A ce titre, elle représente l'unité adéquate pour explorer dans un premier temps l'influence de la variabilité du sol, du climat et des pratiques dans le cadre de l'évaluation environnementale par l'ACV. Les implications de l'échelle n+1 et leur possible prise en compte seront discutées dans le cadre de nos perspectives de travail.

Les principaux résultats de cette thèse sont résumés en figure 5.1 qui présente aussi le plan de progression de cette discussion.

Nous avons tout d'abord réalisé l'état de l'art de l'ACV des produits maraîchers pour nous construire une expertise sur les grands enjeux méthodologiques de l'ACV appliquée à ces produits. Nous avons aborder chacun de ces enjeux dans la réalisation d'une ACV pour le cas d'étude de la tomate en maraichage urbain au Bénin. Nous avons commencé par mettre au point un protocole de collecte de données d'inventaire en réalisant une typologie sur des critères agronomiques et proposé une première méthode d'estimation des émissions au champ (méthode dite par défaut ou DEF) basée sur les meilleures références disponibles dans la littérature. Nous avons ensuite appliqué une approche combinant un bilan d'azote avec l'utilisation du modèle biophysique STICS pour analyser la variabilité des émissions d'azote réactif au champ et mis en évidence 4 principaux paramètres influençant le bilan d'azote à la parcelle. Les jeux de données nouveaux ainsi obtenus ont été mis en œuvre dans une Analyse de Cycle de Vie exhaustive puisqu'elle inclut en plus des catégories d'impact plus classiques telles que changement climatique, eutrophisation, acidification, les impacts liés à l'usage d'eau douce et les différents impacts de toxicité. Au final, quatre grands types de résultats ont été produits : (1) une expertise sur les principaux enjeux relatifs à l'application de l'ACV aux produits maraichers, (2) des développements méthodologiques pour la réalisation d'ACV en contexte tropical pauvre en données, (3) des connaissances agronomiques et environnementales sur les systèmes étudiées et (4) des références relatives aux résultats d'impacts environnementaux qui mettent en évidence les points critiques des systèmes de production. Ces résultats nous ont permis de formuler des recommandations pour de futures ACV de produits maraichers tropicaux et d'identifier des perspectives de recherche communes aux disciplines de l'ACV et de l'agronomie.

Dans cette discussion nous allons donc dans un premier temps revenir sur l'approche de type méta-analyse appliquée à la bibliographie ACV des produits maraichers pour mettre en valeur son potentiel en termes d'expertise et de capitalisation des travaux existants. Ensuite nous reviendrons dans la seconde section sur les moyens mis en œuvre pour prendre en compte la diversité des systèmes de production (section 5.3.1) et intégrer la variabilité en termes de climat, de sol, et de pratiques agricoles (section 5.3.2) au niveau de l'inventaire environnemental de notre système. Dans la troisième section, nous reviendrons sur les résultats d'impact pour les systèmes de culture de tomate en maraichage urbain, en se focalisant sur la portée des résultats et sur les marges de progrès envisageables. Dans la dernière section de cette discussion, seront formulés des recommandations et des axes de recherches communs aux deux disciplines de l'agronomie et de l'évaluation environnementale par l'ACV.

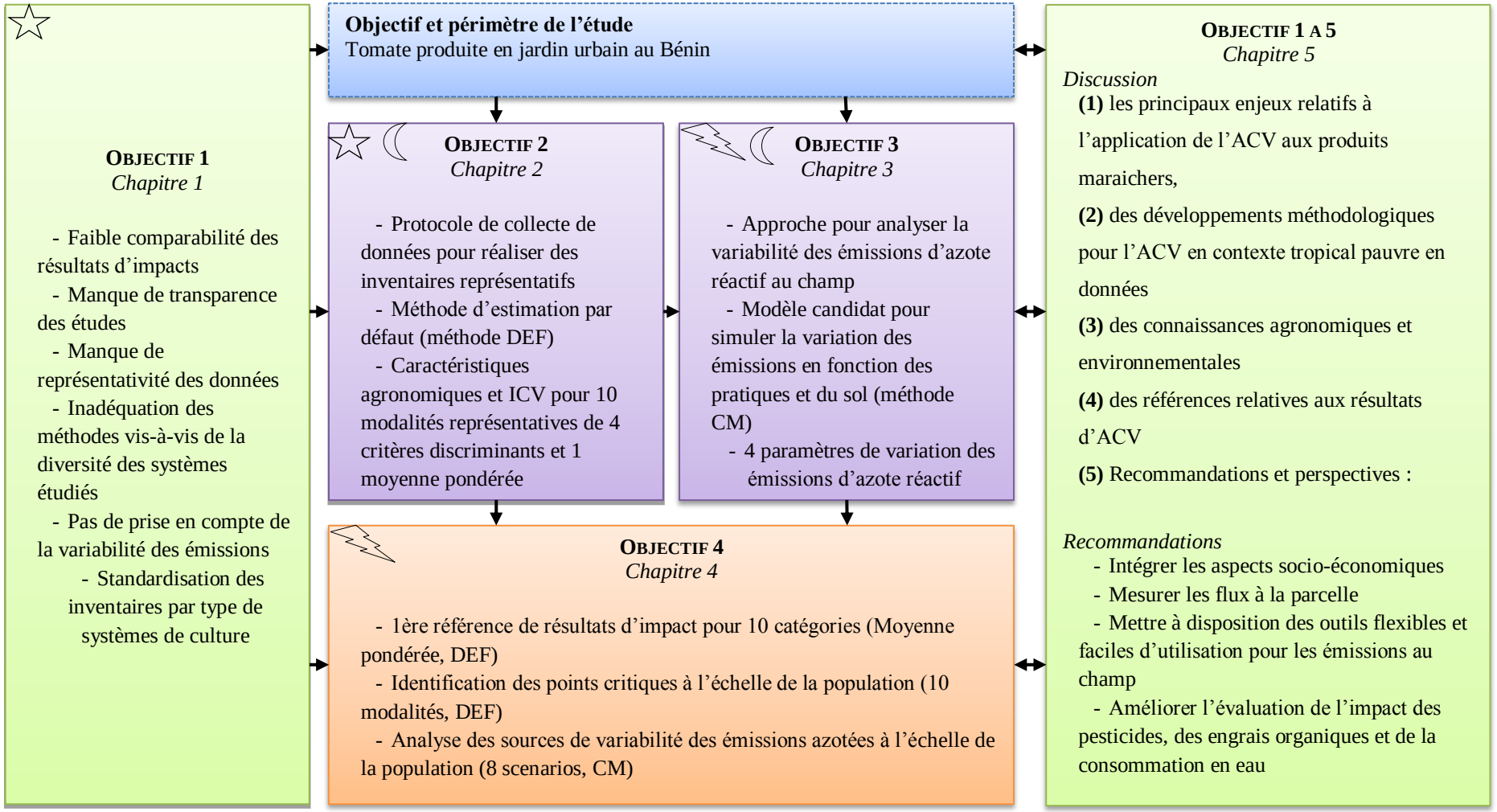


Figure 5.1 Principaux résultats de la thèse et progression de la discussion générale

5.2. Une approche de type méta-analyse pour capitaliser sur le matériau bibliographique

Revenons à l'une des finalités affichées de l'ACV qui consiste à produire des références fiables et comparables concernant les performances environnementales des systèmes de production. L'engouement récent pour l'évaluation environnementale des produits alimentaires a entrainé la multiplication des études ACV dans le domaine agricole. L'approche de type méta analyse offre un moyen efficace de capitaliser sur les études réalisées pour produire des références robustes et pertinentes. Néanmoins, le peu de méta-analyses réalisées dans le domaine agricole s'accordent sur le besoin (1) d'homogénéisation des objets d'étude, (2) d'informations relatives à l'échelle temporelle et spatiale de l'étude et (3) d'informations relatives à la taille de l'échantillon et à l'importance de l'erreur (Bengtsson et al., 2005; Seufert et al., 2012). Il est donc important de remettre en question les moyens à mettre en œuvre pour réaliser des ACV fiables et comparables pour trouver des solutions qui concilient le besoin de standardisation des pratiques d'ACV et de spécificité des outils et des données.

En préalable au travail réalisé sur le cas d'étude Bénin nous avons donc effectué une analyse bibliographique des articles et rapports présentant des ACV « complètes » de produits maraichers (chapitre 1). Dans ce cadre nous avons appliqué une approche de type méta-analyse pour analyser de manière systématique et organisée les pratiques actuelles mises en œuvre en ACV ainsi que leurs résultats. La méta-analyse consiste en l'analyse statistique d'une grande quantité de résultats issus de différentes études dans l'objectif de conforter et mutualiser les résultats existants et aussi de dégager de nouveaux résultats (Glass, 1976). Cette approche nous a d'abord conduits à analyser la qualité des données d'inventaire présentées dans les différents documents disponibles. L'hétérogénéité de ce matériau nous a conduits à proposer plusieurs méthodes d'agrégation des résultats d'ACV basés sur une approche de typologie (à défaut de pouvoir appliquer des méthodes statistiques propres à la méta-analyse).

Dans un premier temps, nous avons mis en évidence l'importante diversité des systèmes de production maraichers et de leurs impacts. Cette diversité résulte d'une augmentation de la demande de fruits et légumes frais tout au long de l'année et se traduit par une augmentation du niveau d'artificialisation des systèmes de production. Nous avons établi qu'une part non négligeable des variations d'impact (en particulier pour le changement climatique) peut être expliquée par un nombre restreint de paramètres clefs (tableau 1.3). Par exemple, nous avons identifié le rendement et les pratiques de fertilisation comme les principales sources de variation des performances environnementales des systèmes de production maraîchers. Nous avons aussi mis en évidence que des paramètres très spécifiques comme l'origine de l'énergie dans le cas de serres chauffées, ou les règles d'allocation associées à l'usage de compost pour des systèmes plein champ pouvaient entrainer des variations non négligeables des résultats.

Nous avons également identifié l'estimation des émissions au champ comme un point critique générant des variations entre les études. En effet, non seulement la mise en œuvre des méthodes est souvent décrite de manière insuffisamment précise, mais en plus les méthodes choisies elles-mêmes présentent des domaines de validité inappropriés et une marge d'adaptabilité insuffisante pour l'évaluation des flux des systèmes maraîchers. Un approfondissement bibliographique sur ces méthodes d'estimation des flux au champ nous a permis de constater qu'aucune des méthodes d'estimation employées dans les études ne peut être appliquée de manière générique à tous les systèmes maraichers (tableau 1.3). Les recommandations formulées dans cette thèse peuvent contribuer à la mise en place de standards propres à l'évaluation environnementale des différents systèmes de culture maraîchers en tant que catégorie de produits plus ou moins homogène.

Finalement, nous avons aussi dû constater une grande variabilité des résultats dont il n'est pas toujours évident de déterminer l'origine. Que celle-ci soit liée à des différences de qualité de données employées ou à des différences réelles entres les systèmes étudiés, le manque de transparence des études rend difficile son analyse. Cette variabilité doit être mieux caractérisée dans l'étape de définition des objectifs et du champ de l'étude. Peu d'études peuvent prétendre présenter un niveau d'information compatible avec les recommandations du guide ILCD (European Commission, 2010) notamment en terme de description de la qualité des données (représentativité et fiabilité). Des informations quantifiées sur la représentativité de l'étude et sur la taille de l'échantillon permettrait d'intégrer cette variabilité dans la méta-analyse et de quantifier leur effet. Par exemple la prise en compte d'un critère « échelle de l'étude » dans la méta-analyse consacré à l'effet de l'agriculture biologique sur la biodiversité (Bengtsson et al., 2005) a montré que la différence entre agriculture biologique et agriculture conventionnelle était plus marquée dans les études à petite échelle.

Plus généralement, ces standards doivent définir les données nécessaires et suffisantes à renseigner pour la réalisation et la publication d'ACV et leur inclusion future dans une approche de type méta-analyse. Ils doivent aussi proposer des méthodes transparentes et flexibles pour intégrer la diversité des systèmes de production et identifier finement les sources de variabilité principales des impacts environnementaux de la catégorie des produits maraîchers. C'est à cette ambition que nos recommandations apportent une première contribution. Nous n'avons pas traité l'ensemble des sources de variation. Notamment la diversité des logiciels et des méthodologies de caractérisation des impacts n'a pas été abordée et pourrait faire l'objet d'une revue bibliographique dédiée.

5.3. Contribution à la réalisation d'inventaires environnementaux

5.3.1. Proposition d'un protocole de collecte des données pour réaliser des inventaires représentatifs de la diversité des systèmes de production

La création d'inventaires représentatifs de la diversité des systèmes de production est l'un des défis majeurs de l'application de l'ACV dans le domaine de l'agriculture. Les méthodes d'agrégation comme la moyenne pondérée (Coltro et al., 2009) ou la scénarisation (Basset-Mens et al., 2007) et les méthodes de désagrégation comme la typologie (Dalgaard et al., 2006) ou la modélisation (Nemecek & Erzinger, 2005) ont montré leur potentiel. Ces tentatives intéressantes ne permettent cependant pas de créer des inventaires dans un contexte où les données sont rares et l'expertise est difficile à mobiliser. Dans cette thèse, la création d'inventaires environnementaux qui prennent en compte la diversité des systèmes de production a reposé sur l'usage de la typologie à dire d'expert pour sélectionner l'échantillon de systèmes à étudier et agréger les données collectées.

Cette démarche avait pour objectif de pallier les fortes contraintes de disponibilités de données dans le contexte d'étude. Elle peut s'appliquer plus généralement dans le cadre d'analyses de systèmes innovants ou de systèmes « de niche » pour lesquels les données génériques ne sont pas valides et les données spécifiques sont insuffisantes. Elle a abouti au développement et à l'application d'une procédure de création d'inventaires présentée dans le chapitre 2 (Figure 2.2). Nous avons ainsi créé des inventaires environnementaux pour 10 modalités issues de 4 critères discriminants pour les systèmes de culture de tomate de contre-saison au Bénin, et pour une moyenne pondérée qui agrège cette diversité sur la base de la répartition des types au sein de la population en fonction de leur surface relative. Ces inventaires présentent une qualité satisfaisante au regard des indicateurs de qualité des données employées en ACV. Sur la base de la matrice des pédigrées développée par Weidema et Wesnæs (1996) et dont l'usage est recommandé en ACV par les guides de bonne pratique récents (European Commission, 2010), on atteint un score maximal de 1/5 pour les critères de fiabilité et de corrélation temporelle et géographique. Seuls les critères de complétude et de corrélation technique présentent des lacunes relatives à la durée de la période de collecte de données (seulement un cycle de culture) et à la répartition des systèmes échantillonnés dans les types (pas suffisamment de système avec irrigation par aspersion et pas de systèmes à cycle tardif).

Nous avons pu relier certaines caractéristiques agronomiques aux systèmes groupés par modalités de la typologie. Aux trois critères initialement sélectionnés : le type de système d'irrigation, l'origine de l'eau et la date de plantation, nous avons ajouté le critère de pression foncière qui repose sur la distribution des types dans chaque commune et le gradient décroissant de pression foncière entre Cotonou et Grand-Popo (la troisième commune de l'étude, Ouidah, présentant un profil intermédiaire). Le critère origine de l'eau d'irrigation a un effet indirect sur la consommation en eau au travers de l'effet type de sol par la présence de sols argileux aux alentours des rivières (l'une des sources d'eau d'irrigation).

L'usage d'insecticides (et plus particulière de la cypermethrine) et de fongicide est plus intense pour les systèmes avec une irrigation manuelle ou ayant recours à l'eau puisée dans une mare, ou avec un cycle de culture précoce ou localisée à Cotonou. La répartition des parcelles au sein de ces modalités se recoupant nous n'avons pu identifier si l'un de ces facteurs avait un effet dominant. Nous avons proposé des hypothèses explicatives qui reposent sur une expertise de terrain mais qui n'ont pas pu être vérifiées par les données techniques collectées (niveau de maitrise technique, accessibilité des intrants, pression parasitaire).

Seule la localisation géographique semble être corrélée avec les pratiques de fertilisation avec des pratiques d'autant plus intenses que le gradient de pression est fort. Les variations de rendement sont plus difficiles à caractériser.

De manière plus globale et pour notre échantillon, nous avons mis en évidence un niveau d'optimisation des systèmes de production imparfait, souvent lié à des conditions de production très contraignantes (pression parasitaire, température élevée, faible accessibilité aux moyens de production) et associé à une faible maitrise technique (faible appui technique, matériel de mesure inadapté, non maitrise des instructions relatives aux intrants). Ce constat explique notre difficulté à caractériser des types contrastés présentant une cohérence globale.

Nous avions envisagé d'autres pistes pour aller plus loin dans l'usage de la typologie pour la réalisation d'inventaires diversifiés et contrastés. L'une d'elle consistait à utiliser les méthodes de typologie quantitative, et notamment celles basées sur l'analyse en composante principale et la classification hiérarchique. Ces méthodes ont démontré leur robustesse dans des contextes où la typologie doit être établie avec peu de connaissances préalables et dans des contextes de grande variabilité (Köbrich et al., 2003). De plus, nous avons identifié le potentiel de cette procédure pour identifier de manière objective des individus représentatifs de la population (appelés parangons) sans avoir recourt à l'agrégation de données qui nécessite des hypothèses et le calcul de moyennes au détriment de la qualité des inventaires. Considérant le nombre important de variables descriptives des systèmes de production, l'application de ces méthodes quantitatives nécessite cependant l'échantillonnage d'un grand nombre d'individus. La typologie présentée par Blazy et al. (2009) réalisée sur 46 variables à partir d'un échantillon de 66 fermes et présentant une forte discrimination des résultats donne un bon exemple de l'ordre de grandeur minimum du ratio entre le nombre d'individus et le nombre de variables étudiées. La taille de notre échantillon n'était pas suffisante et les résultats préliminaires ont montré une faible puissance de discrimination (valeurs propres faibles distribuées sur un grand nombre de dimensions), ce qui nous a poussé à abandonner cette analyse dans notre contexte d'étude.

5.3.2. Proposition d'une approche intégrée pour l'analyse de la variabilité des émissions

La prise en compte de la variabilité des systèmes de production est l'un des défis majeurs de l'application de l'ACV dans le domaine de l'agriculture. La variabilité des pratiques agricoles a été analysée dans plusieurs articles par le biais d'analyses de scénarios (Mouron et al., 2006; Cellura et al., 2012). Ces tentatives intéressantes ne permettent cependant pas de relier l'influence de pratiques agricoles clés comme l'irrigation avec les émissions au champ faute de modèles d'estimation des émissions suffisamment raffinés. D'autre part, les articles traitant de l'analyse de scénarios des impacts aux émissions soulèvent le problème de covariance entre les paramètres impliqués dans les processus d'émission (Payraudeau et al., 2007). L'approche que nous proposons s'intègre dans la continuité des travaux sur l'analyse de scénarios, mais se distingue par une intégration cohérente des paramètres de pratiques et de sol.

Compte tenu de la liste importante de paramètres influents (Tableau 3.1), il était nécessaire de caractériser la complexité dans la gamme de variation du cas d'étude pour identifier un nombre restreint de paramètres sensibles à propager sur les résultats d'ACV. Nous avons donc exploré la variabilité des émissions dans la gamme de parcelles représentatives de la fonction étudiée. Pour cela, nous avons combiné l'approche bilan d'azote avec la modélisation biophysique des flux (chapitre 3, section 3.2.1). Cette combinaison nous a permis d'utiliser de manière cohérente un modèle dynamique à pas de temps journalier dans un cadre d'analyse à l'échelle du cycle de culture. Nous avons identifié 4 paramètres présentant un degré de corrélation élevé avec les variations des composantes du bilan d'azote (Tableau 3.5). Il s'agissait de la dose d'azote, du volume d'irrigation, du pH du sol et de l'humidité du sol à la capacité au champ.

Compte tenu de la corrélation entre les différentes caractéristiques des systèmes de production la mise en œuvre des méthodes de propagation de l'incertitude de type Monte Carlo nécessite le calcul d'indices de covariances entre ces caractéristiques (Payraudeau et al., 2007), pour lesquels nous ne disposons pas d'information a priori dans notre cas d'étude. Nous avons donc réalisé dans un troisième temps des scénarios agronomiques pour propager cette variabilité sur les résultats d'impact. Sur la base de valeurs minimum et maximum de chacun des paramètres au sein de la population, nous avons construit des scénarios représentatifs de conditions réalistes favorables ou défavorables aux émissions dans le domaine de variation de notre échantillon. Pour les paramètres relatifs aux pratiques agricoles, ce travail de scénarisation a nécessité la modélisation à l'échelle journalière (date et fractionnement) de variations de quantités à l'échelle du cycle de culture. Ce changement d'échelle s'est basé sur l'expertise acquise sur les 12 parcelles étudiées. Pour les facteurs relatifs au sol, nous avons dû considérer les relations biophysiques entre les différents paramètres d'entrée du modèle afin de proposer des jeux de paramètres cohérents.

Cette approche nous a permis d'explorer la variabilité des inventaires de manière systématique et transparente. Elle nous a permis d'identifier les principales sources d'incertitude relatives au modèle. Le manque de données pour paramétrer finement le modèle (variété, potentiel de dénitrification, engrais organique), et le manque de données mesurées pour le valider (bilan hydrique et azoté) limitent la portée de ce type de modèle complexe pour raffiner l'estimation des émissions car il réduit sa qualité prédictive (Wallach & Genard, 1998). Par ailleurs, l'incapacité du modèle à reproduire les variations de rendement représente une limite d'ordre conceptuel. Compte tenu du degré de maitrise technique faible des producteurs, il semblerait plus pertinent d'aborder la question du rendement avec des outils probabilistes plutôt que mécanistes. Nous pouvons suggérer comme principaux paramètres (ou ensembles de paramètres) pour établir des relations statistiques plus robustes : la pression parasitaire, la température à la floraison et la connaissance technique des agriculteurs.

Enfin, l'approche bilan nous a permis d'invalider l'hypothèse d'un état stable de la matière organique du sol pour lequel les entrées d'azote seraient équivalentes aux sorties. Il devient nécessaire d'explorer le devenir de l'azote au-delà des limites du système en simulant avec le modèle un grand nombre d'années climatiques. Néanmoins, une année climatique étant composée de plusieurs cycles de cultures d'espèces différentes et présentant des itinéraires techniques propres, cette exploration nécessite des données ou de l'expertise sur l'ensemble de la rotation. Ces limites soulèvent de nouvelles questions scientifiques relatives aux données et aux outils qui feront l'objet d'une analyse complémentaire dans la section perspective de cette discussion.

5.4. Contribution à l'évaluation environnementale de l'agriculture urbaine par la méthode ACV

5.4.1. Impact environnemental de la tomate produite en maraîchage urbain au Bénin

Nous avons réalisé l'analyse environnementale d'un kilogramme de tomate produite en maraîchage urbain au Bénin sur la base de 11 catégories d'impact. L'analyse de la moyenne pondérée nous a permis de formuler pour chaque pratique, quelles sont les composantes critiques des systèmes de culture par rapport à des références de systèmes maraîchers. L'analyse comparative des systèmes regroupés par modalité pour les 4 critères discriminants de notre typologie nous a permis d'identifier les leviers d'action pour réduire l'impact sur l'environnement de la fonction étudiée.
Nos résultats d'ACV sont les premiers pour un système de production de jardin urbain africain d'où la difficulté à comparer avec d'autres références équivalentes. Quand cela était possible (WDP, GWP, AP, et EP), nous avons identifié des scénarios issus de la littérature pour comparer nos résultats d'impact par kilogramme de tomate (chapitre 4, section 4.3.2).

Il ressort qu'un kilogramme de tomate produite en maraîchage urbain présente un potentiel de privation en eau plus élevé que la moyenne des systèmes de production de tomate au Bénin. Il ressort également qu'un hectare de tomate produite en maraîchage urbain présente des potentiels de changement climatique, d'acidification et d'eutrophisation plus élevés que les systèmes maraîchers européens. Nous avons identifié la nature de l'énergie et la quantité d'énergie consommée par volume d'eau prélevé comme les points critiques pour l'impact changement climatique. Nous avons identifié l'effet du climat plus chaud sur le processus de volatilisation et la dose importante d'engrais (notamment d'origine organique) par rapport aux apports totaux d'azote comme les points critiques pour les impacts d'acidification et d'eutrophisation. Nous remarquons qu'une part non négligeable de ces forts impacts repose sur les choix méthodologiques adoptés pour l'estimation des émissions au champ dans la méthode de facteurs d'émission par défaut (DEF).

Enfin, malgré l'absence de références strictement comparables d'un point de vue méthodologique (calcul des émissions et caractérisation des impacts), il ressort qu'un kilogramme de tomate produite en maraîchage urbain présenterait des potentiels d'écotoxicité terrestre et aquatique plus élevés que les systèmes maraîchers européens. Nous avons identifié l'utilisation de molécules avec un potentiel de toxicité particulièrement élevé (des molécules qui sont d'ailleurs non homologués en Europe), les applications systématiques et le surdosage comme les points critiques relatifs au potentiel de toxicité des écosystèmes.

5.4.2. Résultats de l'analyse de scénario : vers un raffinement des méthodes d'estimation au champ

Les effets prépondérants associés au choix du facteur d'émission sur le potentiel d'acidification et d'eutrophisation remettent en question l'usage de méthodes d'estimation des émissions basées sur des facteurs d'émissions peu spécifiques (méthode DEF). Les facteurs d'émission relatifs et génériques sont-ils valides dans nos contextes de production spécifiques ? L'ACV peut-elle apporter des éléments de réponse plus locaux pour limiter l'impact de ces émissions ? Le modèle biophysique STICS a été utilisé pour simuler les émissions pour un scénario de référence (moyenne pondérée) et pour les 8 scénarios agronomiques représentatifs de situations favorables et défavorables aux émissions (chapitre 4, section 4.2.4). L'analyse de ces 8 scénarios nous a permis d'identifier les variations d'impact associées à la variabilité des paramètres influençant le plus les émissions azotées.

En moyenne, le modèle suggère un facteur d'émission de protoxyde d'azote proche de celui proposé par IPCC (2006). Par contre, les taux de volatilisation estimés par le modèle pour nos systèmes sont inférieurs au facteur d'émission trouvé dans la littérature (Bouwman et Van Der Hoek, 1997) ainsi qu'aux mesures dans des contextes comparables (Predotova et al., 2010; Lompo et al., 2012). Le taux de lessivage semblerait lui plus élevée que le facteur générique proposé par IPCC (2006a) ce qui peut s'expliquer par la forte occurrence de sols sableux et les forts volumes d'eau d'irrigation dans notre échantillon.

Nous avons montré que les facteurs d'émission n'étaient pas constants et qu'ils variaient en fonction des volumes d'eau apportés, de la dose d'azote et des paramètres de sol tels que le pH et la capacité au champ (chapitre 4, tableau 17.4) et que ces variations étaient complexes à cause de l'interdépendance des processus biophysiques impliqués. Les variations d'émissions engendrées par les 4 paramètres d'influence étudiés entrainent des variations d'impact non négligeables. La quantité d'eau maximum appliquée à l'échelle de la population entraine un potentiel de changement climatique 3 fois supérieur à la référence. La dose d'azote maximum appliquée à l'échelle de la population entraine un potentiel d'acidification 2,5 fois supérieur à la référence et un potentiel d'eutrophisation 3.2 fois supérieur à la référence. Ces résultats montrent que les milieux sont très sensibles à une augmentation de la dose d'azote du fait que la culture ne valorise pas ce surplus de nutriments. Enfin, l'analyse de scénarios a permis de mettre en évidence des risques de transfert d'impact en fonction des interactions milieu x pratiques. Ainsi une faible capacité au champ permet de réduire la dénitrification et donc la contribution des émissions de N_2O direct au changement climatique. Cependant, une faible capacité au champ augmente aussi le drainage et la volatilisation entrainant une augmentation du potentiel d'acidification, d'eutrophisation et au final une augmentation du potentiel de changement climatique associée à de plus fortes émissions indirectes de N_2O. Ce phénomène est d'autant plus marqué que les cultures sont irriguées et donc que les rendements sont peu sensible à d'éventuels stresses hydriques associés à une faible capacité au champs.

Ces résultats soulignent le besoin de prendre en compte l'irrigation, le pH du sol et la capacité au champ dans l'estimation des émissions azotées. L'usage de modèle biophysiques est une piste prometteuse pour atteindre ce niveau de spécification et des exemples réussis se multiplient dans la littérature (Gabrielle et Gagnaire, 2008; Bessou et al., 2013b; Dufossé et al., 2013). Cependant, la complexité des modèles de culture engendre un besoin important en données de paramétrage et de validation (section 5.3.2 de cette discussion générale) qui limite leur usage dans des contextes de carence en données et en expertise. La nécessité de mettre à disposition des méthodes et des outils adaptés et faciles d'utilisation fait l'objet d'une analyse complémentaire dans la section perspective de cette discussion.

5.4.3. Identification des "hot-spots" à l'échelle de la population

Au Bénin, les producteurs des zones urbaines et péri-urbaines cultivent principalement la tomate en contre saison (de septembre à avril), ce qui implique des pratiques d'irrigation particulièrement intensives (en termes de système d'irrigation et de volumes d'eau apportés) par rapport à d'autres jardins urbains en Afrique de l'Ouest où la tomate est principalement cultivée à partir d'avril en saison plus humide. Cette particularité du Bénin résulte de la production massive de tomate pendant les saisons plus humides en zone rurale. Ces modes de production plus extensifs affectent largement les prix à cette période et rendent la production intensive en jardins urbains beaucoup moins rentable. Les pratiques d'irrigation ont un impact environnemental principalement en termes de potentiel de privation en eau, de potentiel de changement climatique, de toxicité humain, de potentiel de formation photochimique d'ozone et d'écotoxicité marine.

Les performances des pratiques d'irrigation peuvent être améliorées par la promotion des systèmes d'irrigation plus économes qui permet de réduire le volume d'eau appliqué. Dans un second temps, le renouvellement et l'entretien des pompes permettrait de limiter les quantités d'essence consommées par volume d'eau appliqué. Enfin, il pourrait également être intéressant de promouvoir la production au bord des rivières ou des lagunes où les sols présentent une plus grande capacité de rétention de l'eau, ceci en s'assurant que cela ne conduise pas à un transfert de pollution sous la forme de plus grandes émissions de protoxyde d'azote (N_2O) par dénitrification.

Au même titre que les autres jardins urbains d'Afrique de l'Ouest, les systèmes de production de tomate en maraîchage urbain au Bénin présentent des pratiques de fertilisation intensives basées à la fois sur des engrais minéraux (ou synthétiques) et des fertilisants organiques. Ces pratiques de fertilisation ont principalement des impacts en termes de potentiel d'acidification et de potentiel d'eutrophisation.

Nous avons constaté grâce au bilan d'azote à l'échelle du cycle de culture que les flux d'intrants azotés et phosphorés étaient considérablement supérieurs aux exportations par la plante. Compte tenu de la faible concentration en azote des fruits.et du niveau des rendements nos systèmes de culture Béninois exportent peu d'azote comparé à d'autres systèmes maraîchers (chapitre 1). Il est nécessaire d'optimiser les pratiques de fertilisation par rapport aux rendements. Cette optimisation doit prendre en compte la sensibilité particulière de l'urée et de la fiente de volaille à la volatilisation en contexte tropical. La limitation de l'usage des engrais organiques et de l'urée sera d'autant plus utile sur les sols à pH élevé. Cependant, une diminution significative des émissions en azote ne sera permise qu'en alliant une diminution de la dose totale d'engrais à une optimisation des pratiques d'irrigation.

Au même titre que les autres systèmes maraîchers d'Afrique de l'Ouest, les systèmes de production de tomate en maraîchage urbain au Bénin présentent des pratiques de protection des cultures intensives résultant à la fois de l'usage fréquent et en quantités importantes des produits phytosanitaires. Ces pratiques phytosanitaires ont principalement des impacts en termes de potentiels de toxicité des écosystèmes terrestres et d'eau douce, responsables d'une réduction de la qualité des écosystèmes.

Une nette amélioration des performances environnementales peut être atteinte pour nos systèmes par la limitation de l'usage de certaines matières actives comme la cyperméthrine et le terbufos. Nous avons émis l'hypothèse que la pression parasitaire était plus importante pendant le cycle de production précoce. Il serait intéressant d'étudier les variations de pression parasitaire au cours du temps pour promouvoir la culture de tomate pendant les périodes de pression parasitaire moindre. Enfin, la formation et le suivi technique des producteurs sont indispensables pour réduire l'usage de produits de manière systématique et à dose trop élevée.

5.5. Recommandations et perspectives pour améliorer la robustesse des résultats d'ACV en contexte tropical

5.5.1. Intégrer les aspects socio-économiques

Notre approche a consisté à analyser les performances d'un système de culture, c'est-à-dire sans intégrer pleinement les logiques d'exploitation à l'échelle temporelle (rotation) et spatiale (assolement). Ce choix est cohérent avec l'approche produit de l'ACV, d'ailleurs la majorité des études analyse les systèmes à cette échelle (chapitre 1, section 1.3.1). En effet, l'analyse à l'échelle du cycle de culture s'est révélée pertinente pour l'identification de leviers d'action liés aux conditions du milieu, permettant un raisonnement en fonction des interactions milieu x pratiques. Néanmoins, une récente revue bibliographique (Peacock et al., 2011) rapporte que ce choix est souvent une limite majeure des résultats d'ACV et nous ne pouvons qu'être d'accord avec ce constat. Pour notre cas d'étude, nous avons constaté qu'une approche à l'échelle de l'exploitation présente un potentiel explicatif plus fort pour caractériser les systèmes (section 5.3.1 de cette discussion générale). Dans un contexte où les données et l'expertise sont rares, une meilleure compréhension des déterminants des systèmes de production nous aurait permis d'élargir le cadre de notre analyse et de produire des préconisations directement à l'attention du centre de prise des décisions : l'agriculteur. Nous avons également constaté un défaut du bouclage du bilan d'azote à l'échelle du cycle de culture et émis l'hypothèse que ce défaut pouvait être résolu à l'échelle d'une année climatique en prenant en compte les autres cultures de la rotation (section 5.3.2 de cette discussion générale). A ce stade nous ne pouvons qu'encourager fortement les initiatives d'évaluation à l'échelle de l'exploitation.

Dans le cadre de la petite agriculture, notamment en Afrique la ferme est assimilée à un ménage qui représente l'unité de base sur laquelle sont réalisées les activités de production, de consommation et d'éducation. Plusieurs études montrent que ce niveau d'intégration est pertinent pour l'analyse des performances des systèmes de production et de leur impact environnemental (Bidogeza et al., 2009; Tittonell et al., 2010; Sakané et al., 2012). Cependant, la description d'un ménage nécessite la collecte d'un grand nombre de variables caractérisant la structure familiale, les différentes activités économiques, les facteurs de production (travail et capital), ce qui nécessite un grand nombre d'individus enquêtés. Par exemple, la typologie des ménages au Rwanda présenté par Bidogeza et al. (2009) est basée sur un échantillon de 90 fermes. Dans cette perspective, nous avions envisagé d'inclure des paramètres socio-économiques à notre typologie dès le début de la thèse mais ceci n'a pas pu être mis en place faute de moyens suffisants. L'expertise acquise dans le cadre de cette thèse souligne que ce type d'approche nécessite à la fois des compétences spécifiques (notamment en sciences économiques et sociales) et la mise en place d'un protocole de collecte de grande ampleur. Utiliser des modèles à l'échelle de la ferme voire de la région pour analyser l'allocation des ressources est une alternative prometteuse (Parsons et al., 2011; Affholder et al., 2013). Ce type d'approche « top-down » permet à la fois de limiter le nombre de données à collecter, de valoriser l'expertise

existante à l'échelle de l'exploitation et d'intégrer des logiques de stratégie d'exploitation à plus long terme.

Enfin, l'intégration de paramètres socio-économiques au niveau de la définition de l'objet d'étude représente un premier pas vers des démarches intégratrices à l'échelle de l'évaluation des impacts. L'ACV sociale est un domaine de recherche en pleine expansion (Benoît et al., 2010) et son application aux productions horticoles tropicales est pertinente et prometteuse (Feschet et al., 2013).

5.5.2. Collecter des données spécifiques sur les flux à la parcelle

Nos résultats démontrent que l'amélioration de la robustesse des résultats d'ACV ne pourra être possible que par la production de données et de connaissances nouvelles notamment par le biais de mesures biophysiques associées à des outils d'exploration. Pour notre cas d'étude, deux questions précises peuvent être formulées : (1) Quel mode de gestion des engrais azotés permettrait de limiter l'ensemble des émissions d'azote réactif vers l'air et l'eau? (2) Quelles sont les pratiques qui permettraient de réduire le transfert des insecticides en général et des pyrethrinoïdes en particulier vers les eaux de surface et les eaux souterraines? De plus, notre démarche a soulevé des questions de représentativité et de fiabilité qui doivent être intégrées à ces questions scientifiques pointues pour permettre à la connaissance théorique spécifique d'être utilisée de manière pratique dans un cadre plus large d'évaluation des système de production. Par exemple, la problématique des pesticides doit tenir compte des différents niveaux de risque associés aux variations de pression parasitaire, du manque de maitrise technique des producteurs par rapport à l'identification des nuisibles et au dosage des produits et de la toxicité spécifique de chaque molécule. D'autre part, la problématique des engrais azotés est associée à la question des limites du système (délai entre l'apport d'azote et l'émission) et d'allocation des impacts (répartition de l'apport en nutriments au niveau de la rotation). Nous avons montré dans cette thèse que l'analyse du cycle de vie était génératrice de questions de recherche pointues sur le fonctionnement des systèmes de production. L'ACV permet de cibler et de hiérarchiser les lacunes scientifiques relatives à la fonction étudiée. Nous pensons qu'à l'avenir l'ACV mériterait d'être intégrée de manière plus systématique en amont des démarches de définition de projets scientifiques.

5.5.3. Mettre à disposition des méthodes et des outils adaptés

Nous avons exploré la problématique des méthodes et des outils tout au long de cette thèse. Nous avons d'abord insisté sur la nécessité qu'ils soient transparents et accessibles pour optimiser leur utilisation par les praticiens en ACV. Après une analyse plus détaillée et une application dans le cadre du cas d'étude, nous avons montré que ces méthodes et outils gagneraient à être également plus flexibles et plus spécifiques. Enfin, notre expérience relative à l'utilisation du modèle STICS démontre que ces méthodes et outils ne doivent pas être trop gourmands en données et/ou trop complexes à

prendre en main. Reprenons les 3 grandes stratégies possibles mises en évidence par Affholder et al. (2012) dans un article sur la modélisation en agronomie : (1) déduire un modèle conceptuel des connaissances acquises, (2) dériver un modèle simplifié à partir de modèles complexes et (3) identifier et hiérarchiser les causes de variation de la ou des variables d'intérêt et utiliser des méthodes multi variées pour les prédire. La première stratégie n'est pas facilement applicable aux productions maraîchères en contexte urbain compte tenu du manque d'expertise. La dernière stratégie est limitée par la disponibilité en données, néanmoins nous avons montré qu'il est possible de mettre en évidence un nombre restreint de paramètres influents. Cette approche n'est donc pas à mettre de côté dans des domaines qui bénéficient d'une littérature abondante. L'usage d'un modèle complexe validé présente donc le meilleur potentiel pour une utilisation en ACV car il permet de couvrir une grande diversité de systèmes, à différentes échelles spatiales et temporelles.

La première étape consiste à valider un modèle (adéquate) en définissant explicitement son domaine de validité dans le temps et dans l'espace. Pour la diffusion de cet outil deux options sont alors envisageables. La première option consiste à proposer une interface simplifiée tout en gardant l'intégrité du modèle. Cette approche a notamment été utilisée en Australie pour mettre à disposition des agriculteurs le modèle APSIM pour suivre les variations de leur potentiel de rendement et ainsi adapter leur pratique au cours du cycle de production (Hochman et al., 2009). Une base de données est mise à disposition de l'utilisateur pour lui permettre de décrire son système (climat, type de sol, culture), et un nombre restreint de paramètres est suffisant pour spécifier les conditions de production et analyser la variabilité associée à ces paramètres (irrigation, fertilisation, voire même pH). La deuxième option est de simplifier le modèle complexe et de produire ce qu'on appelle un méta-modèle. Britz et Leip (2009) présentent un exemple de méta modélisation (basée sur le modèle DNDC) pour déterminer des facteurs d'émission spécifiques. Cette procédure consiste à réduire le modèle à une fonction de réponse déterminée par un nombre limité de paramètres. Cette approche demande un travail conséquent en amont de la mise à disposition de l'outil car elle nécessite des données supplémentaires pour calibrer et valider le méta modèle. Par contre, ce mode de simplification est ensuite moins coûteux en temps pour l'utilisateur.

5.5.4. Améliorer la prise en compte des pesticides, des engrais organiques et de la consommation en eau

Le besoin en méthodes plus spécifiques largement abordée dans la section précédente sur les émissions azotées est aussi valable pour les émissions de substances phytosanitaires. L'usage de méthodes d'estimation plus fines permettrait d'apporter un niveau supplémentaire d'analyse et de préconisation. Les émissions d'insecticides comme les *pyréthrinoïdes* vers les eaux sont particulièrement préoccupantes à cause de leur persistance et de leur toxicité (Feo et al., 2010). Des outils tels que PestLCI (Dijkman et al., 2012) permettent de raffiner la distribution des émissions dans

les différents compartiments de l'environnement en fonction de paramètres spécifiques de sol et de climat. Cependant, ce type d'approche est souvent limité par un accès restreint aux données nécessaires à l'évaluation et à la validation du modèle, garantissant une utilisation optimum. Enfin, nous n'avons pas pris en compte la présence de résidus de produits phytosanitaires sur les tomates après récolte. De récents développements permettent de quantifier ces résidus (Fantke et al., 2011) pour évaluer leur impact potentiel sur la santé humaine. Cet aspect serait intéressant à intégrer compte tenu de l'absence récurrente dans la pratique de délai entre le dernier traitement et la récolte.
Nous avons montré que l'usage d'engrais organiques est un point critique des performances environnementales de la tomate en maraîchage urbain. L'usage de déchets de ferme et de déchets urbains compostés ou non est une problématique majeure de l'agriculture urbaine (Orsini et al., 2013). Une récente revue bibliographique identifie de nombreux bénéfices du compost au-delà de l'apport en nutriments et propose des moyens pour mieux en tenir compte en ACV (Martínez-Blanco et al., 2013). D'autre part d'autres travaux mettent en évidence des impacts (négatifs) qui ne sont pas toujours pris en compte en ACV comme les émissions avant application (fabrication du compost) (Basset-Mens et al., 2007) ou la dégradation des sols causée par les métaux lourds contenus dans les fertilisants organiques (Anikwe & Nwobodo, 2002; Shao et al., 2012). L'intégration de ces flux (émissions avant application et métaux lourd) et l'impact sur la qualité du sol permettrait d'améliorer la pertinence des résultats d'ACV.

Enfin, l'évaluation du potentiel de privation en eau souligne la nécessité de considérer la localisation des systèmes dans l'analyse de l'impact environnemental des systèmes irrigués. Il renforce également le besoin de raffiner les facteurs de caractérisation. D'une part, nous n'avons pas pu discriminer l'impact des différentes origines de l'eau. Or, le contexte régionale (climat, démographie, topographie) et le contexte local (sensibilité des milieux environnants) infllue largement sur la nature et la gravité des impacts. Au sud Benin par exemple il semble que prélever de l'eau dans un cours d'eau est moins impactant que de prélever de l'eau dans la nappe qui est également l'unique ressource en eau pour les consommations domestiques. Un niveau de spécification plus élevé aurait renforcé l'intérêt de privilégier les zones de production en bord de rivière ou de lagune. D'autre part le manque de prise en compte de la saisonnalité dans la caractérisation de la rareté en eau est une limite importante dans nos contextes où l'eau peut être rare en saison sèche puis en excès en saison des pluies.

5.6. Conclusions

Nous insisterons pour conclure sur le fait que l'ACV est un outil de connaissance et de compréhension. Associé à une analyse agronomique, elle nous a permis de souligner 3 principes importants à retenir pour la réalisation d'ACV agricoles :

1. la représentativité des inventaires dépend de la qualité de prise en compte de la diversité des caractéristiques agronomiques et socio-économiques des systèmes de production et de leur niveau d'optimisation,
2. des indicateurs agronomiques simples tels que les bilans d'azote et les indices de traitements en pesticides permettent un éclairage complémentaire efficace aux résultats d'impacts environnementaux,
3. la prise en compte des processus biophysiques dans l'estimation des émissions d'azote réactif augmente les performances prédictives de l'ACV et sa capacité à identifier d'éventuels transferts de pollution.

Nous avons commencé cette discussion en soulignant le besoin de standardisation. Nous la terminons en proposant des pistes de recherche très pointues. La méthodologie ACV propose une procédure normalisée, mais c'est l'utilisateur qui définit le niveau de résolution. A nous chercheurs de proposer des moyens efficaces pour repousser les limites de la connaissance et pour gagner en performance de prédiction. Il reste du ressort des utilisateurs de bien poser leurs questions. J'espère que ma thèse aura contribué à établir un dialogue fructueux entre agronomes et « ACV'istes » en mettant en évidence les enjeux partagés.

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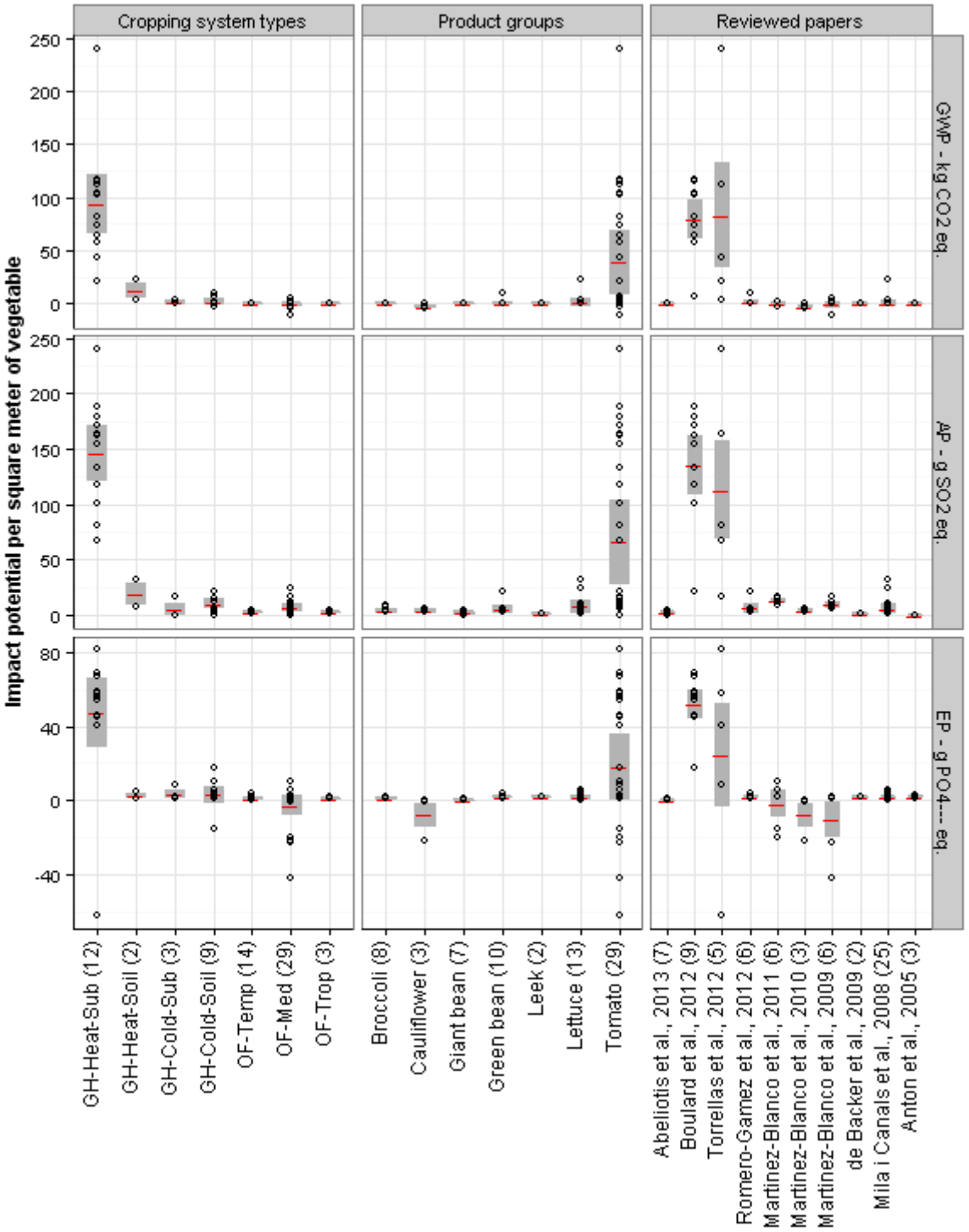
ANNEXES



Paniers utilisés pour la récolte des tomates, sur la route entre Grand-Popo et Cotonou, Bénin

Annex 1. Global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP) expressed per square meter for a vegetable crop cycle from cradle-to-harvest for the 72 systems aggregated per cropping system type, product group and reviewed paper.

Figures in brackets correspond to the number of systems in the group. Grey boxes correspond to error bars centered on the mean of the group (red dash)



Annexe 2. Liste des producteurs de tomate de contre saison pour la saison 2011-2012 sur les communes de Cotonou, Ouidah et Grand-Popo

Cette liste a été établie en collaboration avec les Centres Communaux de Promotion Agricole (CeCPA) en charge de l'accompagnement des producteurs. Le stade correspond aux observations et prévisions réalisées en Octobre 2012 (JAR/SAR/MAR= Jour/semaine/mois après repiquage). Dans la colonne « Traçabilité » sont identifiés les problèmes de traçabilité dues à un stade trop avancé de la culture (A) et à l'incertitude associée aux systèmes les plus tardifs (T). Les croix dans la colonne « Etat sanitaire » identifie les parcelles qui ont déjà subi des attaques parasitaires à la date de l'établissement de cette liste.

Ville	Nom du producteur	Surface m²	Variété	Précocité	Système d'arrosage	Source d'eau	Localisation du site	Stade *	Tracabilité	Etat sanitaire	Type	Echan -tillon
Cotonou	SOGLOHOUN Léonard	216	Mongal	Précoce	Mannuel	Dépression	PLM	4JAR			1	1
Cotonou	HOUNTHO Faustin	234	Mongal	Précoce	Mannuel	Dépression	PLM	Pépinière			1	2
Ouidah	ALLIANCE Zodoté	3200	Akli.	Précoce	Mannuel	Dépression	Mahuclo	15JAR			1	3
Cotonou	DANSOU Dominique	492	Mongal	Précoce	Mannuel	Dépression	PLM	1MAR	A		1	
Cotonou	AMOUSSOU Hamadou	84	Mongal	Précoce	Mannuel	Dépression	Houéyiho	5JAR			1	
Cotonou	DANFADJI	252	Mongal	Précoce	Mannuel	Dépression	PLM	Floraison	A		1	
Cotonou	SEKANDE Bathélémy	702	Mongal	Médian	Mannuel	Dépression	PLM	Pépinière			2	
Cotonou	SANTOS Victor	1356	Caraibe	Précoce	Mannuel	Forage + pompe + reservoir	Fidjrossè	Pépinière			3	4
Ouidah	TONOUKOIN Arsène	3200	Akli.	Précoce	Mannuel	Forage + pompe + reservoir	Ahozon	10JAR		x	3	
Cotonou	ADIMONSSO Félix	294	Tropimech	Précoce	Mannuel	Forage	COMAES	Maturité	A		3	
Ouidah	BADJISSI Emile	800	Mongal	Précoce	Mannuel	Forage	Mahuclo	Repiquage		x	3	
Cotonou	AMOUSSOU Jean Claude	360	Mongal	Médian	Mannuel	Forage	Akpakpa Dodomè	Pépinière		x	4	
Cotonou	GBESSEMEHLAN Antoine	405	Mongal	Médian	Mannuel	Forage	Akpakpa Dodomè	Pépinière		x	4	

Ville	Nom du producteur	Surface m²	Variété	Précocité	Système d'arrosage	Source d'eau	Localisation du site	Stade	Tracabilité	Etat sanitaire	Type	Echan -tillon
Cotonou	HOUNSSOU Vincent	405	Mongal	Médian	Mannuel	Forage	Akpakpa Dodomè	Pépinière		x	4	
Cotonou	KPATENON Sonia	405	Mongal	Médian	Mannuel	Forage	Akpakpa Dodomè	Pépinière		x	4	
Ouidah	AGBO Inès	800	Mongal	Médian	Mannuel	Forage	Ouidah Colas	Pépinière		x	4	
Grand Popo	GOUFLE Théophile	1000	Mongal	Précoce	Tuyau	Forage + pompe	Hounsoukouè	Repiquage			5	5
Grand Popo	EWE Théophile	360	Mongal	Précoce	Tuyau	Forage + pompe	Yodocondji	10JAR		x	5	
Ouidah	YEHOUENOU René	5000	Mongal	Précoce	Tuyau	Forage + pompe	Houdjava	10JAR		x	5	
Ouidah	DOUKPO Jean	5000	Mongal	Précoce	Tuyau	Forage + pompe	Houdjava	15JAR		x	5	
Grand Popo	DJIMEZO Franck	440	Mongal	Précoce	Tuyau	Forage + pompe	EWE-CONDJI	21JAR	A		5	
Ouidah	FAGBOHOUN Janvier	7000	Mongal	Précoce	Tuyau	Forage + pompe	Houdjava	2JAR		x	5	
Ouidah	DOUKPO Augustin	4400	Mongal	Précoce	Tuyau	Forage + pompe	Houdjava	Repiquage		x	5	
Ouidah	NOUWATIN Gervais	1600	Akli.	Précoce	Tuyau	Forage + pompe	Ahozon	Repiquage		x	5	
Ouidah	TONATO Sylvain	800	Mongal	Précoce	Tuyau	Forage + pompe	Ouidah Centre	Repiquage		x	5	
Ouidah	YEHOUENOU Paulin	5000	Mongal	Précoce	Tuyau	Forage + pompe	Houdjava	Repiquage		x	5	
Ouidah	GUEZODJE Charles	4400	Mongal	Médian	Tuyau	Forage + pompe	Akadjamè	Pépinère Novembre			6	6
Ouidah	ADJANOHOUN Elie	6200	Mongal	Médian	Tuyau	Forage + pompe	Houdjava	Pépinière Novembre			6	7
Ouidah	KPATENON Barthélémy	5520	Mongal	Médian	Tuyau	Forage + pompe	Houdjava	Pépinière Décembre			6	8
Grand Popo	EGBEMIKPO Michèle	400	Mongal	Médian	Tuyau	Forage + pompe	EWE-CONDJI	Pépinière Novembre			6	9
Cotonou	TCHIBOZO Maurice	90	Petomech	Médian	Tuyau	Forage + pompe	Houéyiho	Pépinère			6	

Ville	Nom du producteur	Surface m²	Variété	Précocité	Système d'arrosage	Source d'eau	Localisation du site	Stade *	Tracabilité	Etat sanitaire	Type	Echan -tillon
Grand Popo	ADANGBEDOU Daniel	2560	Mongal	Tardif	Tuyau	Forage + pompe	Yodocondji	Pépinière Février	T		7	
Grand Popo	KPODO Régina	3584	Mongal	Tardif	Tuyau	Forage + pompe	Yodocondji	Pépinière Janvier	T		7	
Grand Popo	KASSA Félix	5000	Mongal	Tardif	Tuyau	Forage + pompe	Yodocondji	Pépinière Mars	T		7	
Grand Popo	TOSSOU Codjo	20000	Mongal	Médian	Tuyau	Rivière + pompe	Onkouihoué	Pépinère Décembre			8	10
Ouidah	AGBOGANON Ignace	4400	Mongal	Médian	Tuyau	Rivière + pompe	Akadjamè	Pépinère			8	11
Grand Popo	TOSSOU Dodji	30000	Mongal	Tardif	Tuyau	Rivière + pompe	Onkouihoué	Pépinière Janvier	T		9	
Cotonou	GNAMBODE Francis	450	Mongal	Précoce	Asperseur	Forage + pompe	Akpakpa Dodomè	Pépinière			10	12
Grand Popo	ZINGAN Symphorien	4000	Mongal	Précoce	Asperseur	Forage + pompe	EWE-CONDJI	21JAR	A		10	
Cotonou	KINHOUDE Léonard	1614	Tropimec h	Précoce	Asperseur	Forage + pompe	COMAES	3SAR	A		10	

Annexe 3. Protocole d'enquête pour la collecte des données qualitatives par les conseillers auprès des 12 agriculteurs sélectionnés

FICHE COLLECTE PARCELLE – Stade pépinière et repiquage

Date visite : ……	Nom enquêteurs : ……	Localité : ……
N° Parcelle : ……	Nom& Prénom Producteur : ……	Variété : ……

STADE PEPINIERE

Date début	
Date fin	

Mise en place		
	Nom	Quantité
Variété		
Outils		
Opérateur		

Fertilisation		
Fréquence		
	Nom	Quantité
Produits		
Outils		
Opérateur		

Irrigation		
Fréquence		
	Nom	Quantité
Mode d'apport		
Opérateur		

Protection phytosanitaire		
Fréquence		
	Nom	Quantité
Produits		
Outils		
Opérateur		

REPIQUAGE

Date début	
Date fin	

Repiquage		
	Nom	Quantité
Variété		
Outils		
Opérateur		

Si apport au repiquage :

Fertilisation		
	Nom	Quantité
Produits		
Outils		
Opérateur		

Irrigation		
	Nom	Quantité
Mode d'apport		
Opérateur		

Protection phytosanitaire		
	Nom	Quantité
Produits		
Outils		
Opérateur		

FICHE COLLECTE PARCELLE- Stade culture

Date visite : ……. Nom enquêteurs : ……..…….……. Localité : ……..……..

N° Parcelle[2] : …….. Nom& Prénom Producteur : ……..……..……. Variété : ……..……..

	Fertilisation 1		Fertilisation 2	
Date/Période				
Objectif (Croissance, fructification, carence…)				
Fréquence sur la période				
	Nom	**Quantité**	**Nom**	**Quantité**
Produits				
Outils d'épandage				
Opérateur (producteur, MOF[3], MOS, autre)				

	Traitement phyto 1	Traitement phyto 2
Date/Période		
Cible (maladie, insectes, mauvaises herbes…)		
Fréquence sur la période		

[2] Faire précéder le numéro de parcelle de celui de l'exploitant (Cf liste)

[3] MOF= Main d'œuvre Familiale, MOS= Main d'Œuvre Salariée, autre= main-d'œuvre ponctuelle, entraide voisins…(à préciser)

	Traitement phyto 1		Traitement phyto 2	
	Nom	**Quantité**	**Nom**	**Quantité**
Produits				
Outils de pulvérisation				
Opérateur (producteur, MOF, MOS, autre)				

	Irrigation 1		Irrigation 2	
Date/Période				
Fréquence sur la période				
	Nom	**Quantité/Durée**	**Nom**	**Quantité/Durée**
Mode d'apport				
Opérateur (producteur, MOF, MOS, autre)				

	Récolte 1		Récolte 2	
Date/Période				
Fréquence sur la période				
	Nom	**Quantité**	**Nom**	**Quantité**
Outils de collecte/ contenant				
Opérateur (producteur, MOF, MOS, autre)				
Tomate récolté				

Annexe 4. Liste des producteurs de tomate de contre saison pour la saison 2011-2012 sur les communes de Cotonou, Ouidah et Grand-Popo

Résumé des données - Stade pépinière

Parcelles ID	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Date de semis	1/9/11	2/10/11	5/9/11	2/9/11	9/10/11	1/11/11	1/11/11	1/12/11	5/11/11	11/12/11	17/10/11	17/10/11
Durée pépinière (jours)	38	40	30	63	21	20	20	13	30	25	31	37
Surface (m^2)	2	1	1.8	12	12	6	6	2	11	21.28	7	1.5
Densité ($Graines.m^{-2}$)	714	1429	3175	1190	1190	4762	2905	3571	1299	671	1741	952
Apport fiente (nbre)	1	1	1	1	1	1	1	1			1	
N ($kg.ha^{-1}$)	1319	2639	1233	258	486	530	608	817			1808	
K_2O ($kg.ha^{-1}$)	905	1811	847	177	333	364	417	560			1241	
P_2O_5 ($kg.ha^{-1}$)	1056	2113	988	206	389	424	487	654			1448	
Apport engrais minéral (nbre)	2											
N ($kg.ha^{-1}$)	839											
K_2O ($kg.ha^{-1}$)	213											
P_2O_5 ($kg.ha^{-1}$)	213											
Jour irrigués (nbr)	37	39	29	31	20	19	18	12	29	23	30	36
Volume ($m^3.ha^{-1}$)	6383	11700	4833	1550	11111	950	750	27000	22594	1087	32143	1440
Traitement fongicide (nbr)			1		4		1		1	2	1	
TFI			3.0		7.6		2.5		1.1	2.7	1.0	
Traitement insecticide (nbr)	4	1	2	1	2		1				1	
TFI	9.5	1.7	1.8	2.7	13.3		1.7				1.6	
Traitement nématicide (nbr)						1	1				1	
TFI						7.1	5.8				3.1	

Résumé des données - Stade culture

Parcelles ID	1	2	3	4	5	6	7	8	9	10	11	12
Date de repiquage	9/10/11	11/11/11	5/10/11	4/10/11	30/10/11	21/11/11	21/11/11	14/12/11	5/12/11	5/1/12	17/11/11	23/11/11
Surface pépinière utilisée	100%	100%	100%	25%	100%	33%	100%	100%	100%	100%	16%	100%
Durée culture (jours)	80	64	80	79	92	82	36	91	83	79	57	67
Surface (m^2)	306	126	921.5	546	576	168.72	1914.9	759.83	963	2200	895.2	196
Densité ($Plants.m^{-2}$)	3.85	5.32	1.81	2.74	7.42	3.82	2.40	3.30	5.67	3.34	3.46	3.67
Apport fiente (nbre)	3	2	1	2		2		2			2	1
N ($kg.ha^{-1}$)	1045	64	117	473		563		778			450	558.756
K_2O ($kg.ha^{-1}$)	717	44	81	325		387		534			309	383.46
P_2O_5 ($kg.ha^{-1}$)	836	51	94	379		451		623			360	447.37
Apport engrais minéral (nbre)	5	5	1	7	1	3	2	1	1	5		2
N-Urea ($kg.ha^{-1}$)	33	146	0	43	0	0	0	0	59	202		0
N-NPK (kg.ha-1)	107	104	19	25	22	235	58	0	0	72		211
K_2O ($kg.ha^{-1}$)	107	104	38	25	22	235	58	0	0	72		346
P_2O_5 ($kg.ha^{-1}$)	107	104	38	25	22	235	58	0	0	72		211
Jour irrigués (nbr)	45	48	45	78	69	38	23	64	66	27	50	65
Volume ($m^3.ha^{-1}$)	7200	6360	586	10400	5318	4515	919	10581	7048	771	3759	2570
Traitement fongicide (nbr)	7	1	2		4	2		1		3		3
TFI	7.5	2.0	0.6		2.3	1.2		0.4		3.5		3.8
Traitement insecticide (nbr)	7	2	2	10	3	3	2	1		4	4	3
TFI	14.7	3.7	4.3	25.1	8.8	12.8	1.5	0.6		10.5	5.2	2.8
Traitement nématicide (nbr)						1				1	1	1
TFI						6.1				0.7	7.9	4.9
Récolte (nbre)	1	6	1	3	4	1	1	4	4	5	1	1
Date dernière récolte	28/12/11	14/1/12	24/12/11	22/12/11	30/1/12	11/2/12	27/12/11	14/3/12	26/2/12	24/3/12	13/1/12	29/1/12
Rendements cumulés ($kg.ha^{-1}$)	4902	12452	8	7875	21163	127	0	5662	1703	4498	2710	0

Détail des données - Protection phytosanitaire, stade culture

Parcelles ID	1	2	3	4	5	6	7	8	9	10	11	12
Fongicides												
Cuivre												
Nombre traitements										1		
TFI										0.9		
Mancozèbe												
Nombre traitements			2		2	1		1		1		
TFI			0.6		1.9	0.5		0.4		1.8		
Manèbe												
Nombre traitements	7	1										1
TFI	7.5	2.0										2.13
Souffre												
Nombre traitements					2							
TFI					0.4							
Thiophanate-méthyl												
Nombre traitements						1				1		2
TFI						0.7				0.8		1.7
Insecticides												
Chlorpyrifos-éthyl												
Nombre traitements			2							1		
TFI			4.3							2.3		
Cypermethrine												
Nombre traitements					1	2				1		
TFI					5.2	7.9				4.5		
Deltamethrine												
Nombre traitements						1					1	
TFI						4.9					1.9	

Parcelles ID	1	2	3	4	5	6	7	8	9	10	11	12
Insecticides (Suite)												
Dimethoate												
Nombre traitements							2				2	
TFI							1.5				2.2	
Dimethoate / Cypermethrine												
Nombre traitements	7			5	2			1				3
TFI	14.7			15.6	3.6			0.6				2.8
Lambda cyalothrine												
Nombre traitements		2								2	1	
TFI		3.7								3.6	1.1	
Spinozad												
Nombre traitements				5								
TFI				9.5								
Nematicide												
Carbofuran												
Nombre traitements												1
TFI												4.9
Fenamiphos												
Nombre traitements										1		
TFI										0.7		
Terbufos												
Nombre traitements						1					1	
TFI						6.1					7.9	

Description des produits phytosanitaires utilisés

Nom commercial	Producteurs utilisateurs (nbr)	Cible	Formulation	Substance Active 1 (SA1)	[SA1] ($g.kg^{-1}$ / $.L^{-1}$)	Substance Active 2 (SA2)	[SA2] ($g.kg^{-1}$ / $.L^{-1}$)	Dose homologuée (kg / $L.ha^{-1}$)
Cydim	23	Insectes	Liquide	Dimethoate	400	Cypermethrine	36	0.7
Cypercal	6	Insectes	Liquide	Cypermethrine	50			0.5
Lazer	5	Insectes	Liquide	Spinozad	480			0.15
Lambda Finer 2,5EC	4	Insectes	Liquide	Lambda cyalothrine	25			0.5
Dimex	3	Insectes	Liquide	Dimethoate	400			0.5
SUMITEX 40 EC	3	Insectes	Liquide	Dimethoate	400			0.5
SUNPYRIFOS 48%EC	2	Insectes	Liquide	Chlorpyrifos-éthyl	480			0.5
Decis	2	Insectes	Liquide	Deltamethrine	15			0.6
PACHA 25 EC	2	Insectes	Liquide	Lambda cyalothrine	15	Acetamiprid	10	1
WREKO 2.5EC	2	Insectes	Liquide	Lambda cyalothrine	25			0.5
Dursban 4E	1	Insectes	Liquide	Chlorpyrifos-éthyl	480			0.5
Manate	9	Maladies	Solide	Manèbe	800			2
TOPSIN-M 70 WP	7	Maladies	Solide	Thiophanate-méthyl	700			1
MANCOZEB 80% WP	4	Maladies	Solide	Mancozèbe	800			2
Ridomil plus	3	Maladies	Solide	Cuivre	600	Métalaxyl	120	20
Sulfa 80WDG	3	Maladies	Solide	Souffre	800			10
FOKO	2	Maladies	Solide	Mancozèbe	800			2
CHAMPION	1	Maladies	Solide	Cuivre	770			10
Kocide 2000	1	Maladies	Solide	Cuivre	538			2.5
Conti-Zeb "5" 80% WP	1	Maladies	Solide	Mancozèbe	800			1
Dithane-M	1	Maladies	Solide	Mancozèbe	800			2
Contitop 70% WP	1	Maladies	Solide	Thiophanate-méthyl	700			1
Nématue 10G	4	Nématodes	Solide	Terbufos	100			2.2
Carbofuran 5G	2	Nématodes	Solide	Carbofuran	50			10
Némacur 10G	1	Nématodes	Solide	Fenamiphos	100			100

Matériel d'irrigation

ID parcelles	4	5	6	7	8	9	10	11	12
Débit au niveau de la pomme d'arrosage ($L.h^{-1}$)		2000	1800	1500	3000	2571	3600	3000	
Debit au niveau de la pompe ($L.h^{-1}$)	1800								1800

Profondeur source d'eau (m)	7	7	13	12	12	7	0	0	9
Consomation essence réelle ($L.h^{-1}$)	1.0	3.0	1.0	1.0	2.0	1.0	1.0	1.5	1.0

Conduite d'eau (poids kg*)	37	78	39	31	259	222	217	107	77
Durée de vie des conduites (h)	2208	4944	4410	5570	7152	3240	3402	7820	108
Tuyau flexible (poids kg*)	-	9	9	18	18	18	18	9	-
Durée de vie des tuyaux (h)	-	3708	4410	4456	3576	2592	4536	6256	-
Asperseurs (poids kg*)	-	-	-	-	-	-	-	-	4
Durée de vie des asperseurs (h)	-	-	-	-	-	-	-	-	2025
Pompe (poids kg*)	27	27	27	27	27	27	27	27	27
Durée de vie de la pompe (h)	3312	9888	2646	3240	1788	6480	5670	9384	1296

*Données utilisées pour estimer les poids:
Conduite d'eau: 5.6 kg pour 4m
Tuyau flexible =5.26 kg pour 30m
Asperseur = 0.16 kg par unité

Annexe 5. Exemple des données complètes collectées pour l'itinéraire technique pour la parcelle 6

ID parcelle: 1
Producteur: SOGLOHOUN
Localisation: Cotonou (PLM)
Conseiller: ODRADO Clémence
Système d'irrigation: Manuel **Type:** 1
Origine de l'eau: Depression
Position du cycle: Précoce
Surface pépinière: 2 m² *La totalité utilisé pour le repiquage*
Surface repiquée: 372 m² *51 planches de 6m²*

	Date	Dose	Unité	Materiel
SEMIS	01/09/2011	714	grains.m^{-2}	Houe, rateau
IRRIGATION	Journalière	173	$m^3.ha^{-1}.j^{-1}$	Arrosoir
FERTILISATION				
Fiente	14/09/2011	43	$t.ha^{-1}$	Epandage manuel
NPK 16 16 16	21/09/2011	1329	$kg.ha^{-1}$	Arrosoir
Urée	21/09/2011	1363	$kg.ha^{-1}$	Arrosoir
PROTECTION				
Cydim	01/09/2011	0.7	TFI	Pulvérisateur
Cydim	07/09/2011	0.7	TFI	Pulvérisateur
Cydim	14/09/2011	0.7	TFI	Pulvérisateur
Cydim	21/09/2011	0.7	TFI	Pulvérisateur
REPIQUAGE	09/10/2011	3.9	plants.m^{-2}	Houe, rateau, fil
IRRIGATION	Journalière > 23/11/2011	80	$m^3.ha^{-1}.j^{-1}$	Arrosoir
FERTILISATION				
NPK 16 16 16	16/10/2011	52.1	$kg.ha^{-1}$	Arrosoir
Urée	16/10/2011	71.2	$kg.ha^{-1}$	Arrosoir
Fiente	19/10/2011	11	$t.ha^{-1}$	Epandage manuel
Fiente	29/10/2011	11	$t.ha^{-1}$	Epandage manuel
NPK 16 16 16	03/11/2011	205.3	$kg.ha^{-1}$	Capsule au pied des plantes
NPK 16 16 16	13/11/2011	205.3	$kg.ha^{-1}$	Capsule au pied des plantes
Fiente	16/11/2011	11	$t.ha^{-1}$	Epandage manuel
NPK 16 16 16	20/11/2011	205.3	$kg.ha^{-1}$	Capsule au pied des plantes

	Date	Dose	Unité	Materiel
PROTECTION				
Cydim	23/10/2011	2.37	TFI	Pulvérisateur
Cydim	06/11/2011	2.3	TFI	Pulvérisateur
Cydim	20/11/2011	2.3	TFI	Pulvérisateur
Cydim	10/12/2011	1.5	TFI	Pulvérisateur
Cydim	30/10/2011	2.3	TFI	Pulvérisateur
Cydim	13/11/2011	2.3	TFI	Pulvérisateur
Cydim	01/12/2011	1.5	TFI	Pulvérisateur
Manate	30/10/2011	0.8	TFI	Pulvérisateur
Manate	13/11/2011	0.8	TFI	Pulvérisateur
Manate	01/12/2011	1.7	TFI	Pulvérisateur
Manate	23/10/2011	0.8	TFI	Pulvérisateur
Manate	06/11/2011	0.8	TFI	Pulvérisateur
Manate	20/11/2011	0.8	TFI	Pulvérisateur
Manate	10/12/2011	1.7	TFI	Pulvérisateur
RECOLTE	28/12/2011	4902	$kg.ha^{-1}$	Manuelle

BILAN	**Pepinière**	**Culture**	
Fertlisation organique			
N	1319	1045	$kg.ha^{-1}$
P_2O_5	905	717	$kg.ha^{-1}$
K_2O	1056	836	$kg.ha^{-1}$
Fertlisation minérale			
N	839	140	$kg.ha^{-1}$
P_2O_5	213	107	$kg.ha^{-1}$
K_2O	213	107	$kg.ha^{-1}$
Protection			
Insecticides	9.5	14.7	TFI
Nematicides	0	7.5	TFI
Fongicides	0		TFI
Irrigation			$m^3.ha^{-1}$
Rendements cumulés	>28/12/2011	4902	$kg.ha^{-1}$

Informations complémentaires

Fiente N = 3.06%, P_2O_5=2.45%,K_2O=2.1%

Cydim Cypermethine = 36 $g.L^{-1}$, Dimethoate = 400 $g.L^{-1}$
Dose de reference = 0.7 $L.ha^{-1}$

Manate Manèbe = 800 $g.kg^{-1}$; Dose de reference = 2 $kg.ha^{-1}$

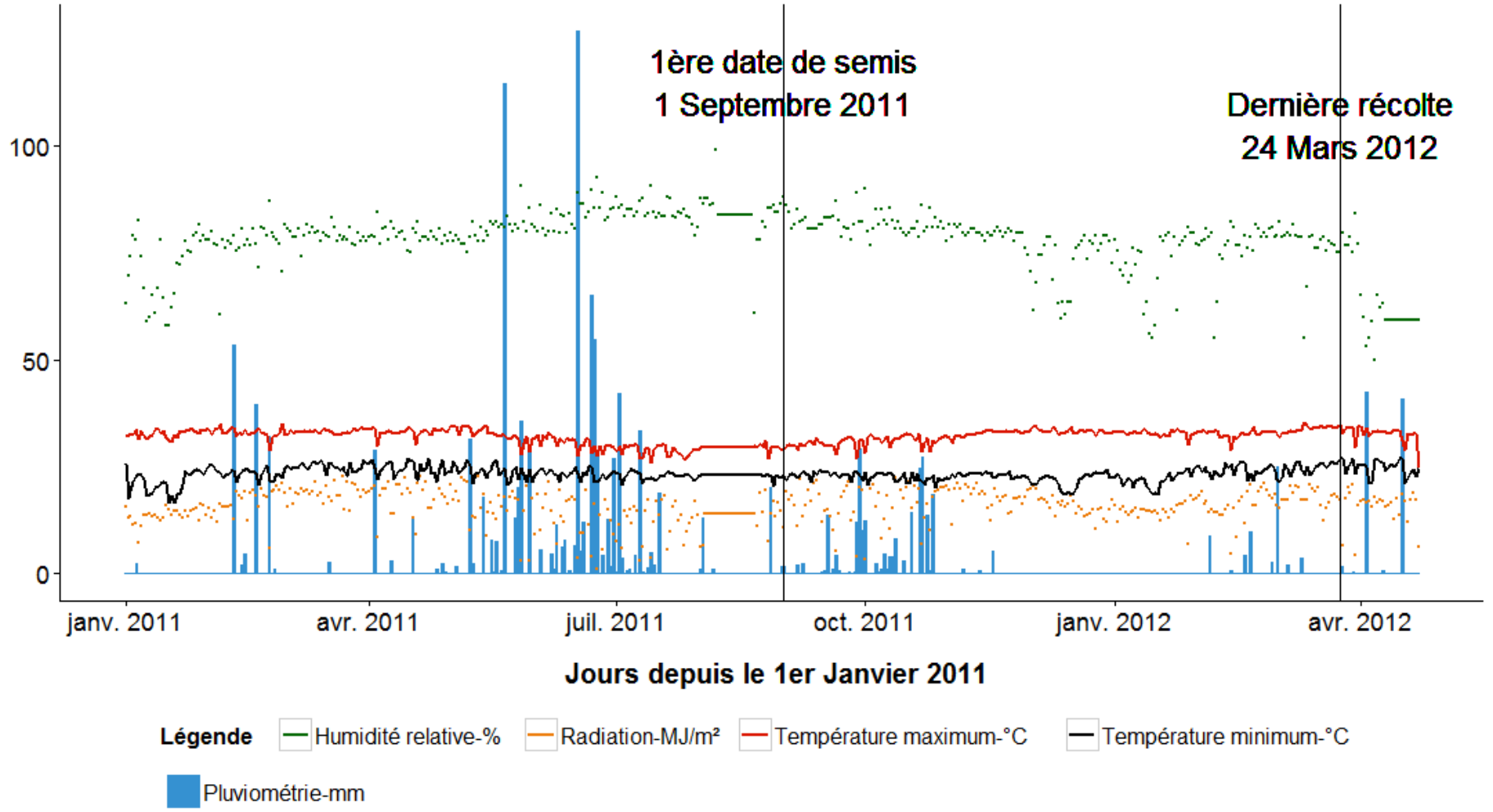
Annexe 6. Résultats des analyses de sol pour les 12 parcelles échantillonnées

Laboratoire: Université d'Abomey-Calavi, Faculté des Sciences Agronomiques
Date: 03/01/2012

	Unit	1	2	3	4	5	6	7	8	9	10	11	12	WM
Granulométrie														
Limons	%	0.75	1	1.75	1.5	1.75	1.25	5	0.75	1.25	12.75	2	0.75	3.26
Argile *	%	7.75	1.75	13	15	4.5	2	9.75	7.75	0.75	65	2	3.25	13.41
Sables	%	91.5	97.25	85.25	83.5	93.75	96.75	85.25	91.5	98	22.25	96	96	83.33
Densité apparente *	g.cm-3	1.43	1.23	1.2	1.41	1.51	1.48	1.34	1.35	1.48	1.27	1.48	1.16	1.412
Humidités caractéristiques*														
Capacité maximale de rétention	%	13.4	13.4	13.6	10.6	10.8	13.0	16.1	24.7	9.6	61.6	18.6	11.4	20.1
Point de fletrissement permanent	%	2.8	2.7	5.2	3.1	2.0	3.5	4.0	4.2	1.7	33.3	3.4	2.2	7.0
C/N *	-	15.2	20.9	30.4	19.0	16.8	21.8	21.8	22.2	23.5	20.6	17.9	19.1	19.1
Azote total	%	0.064	0.050	0.039	0.059	0.042	0.048	0.064	0.064	0.036	0.148	0.070	0.045	0.064
NH4	ppm	4.5	3.8	4.4	2.3	2.2	7.3	1.5	4.6	2.9	3.1	3.1	2.2	2.9
NO3	ppm	13.5	9.1	8.8	6.1	13.3	8.8	9.0	10.7	15.8	13.9	10.8	10.3	11.8
Azote organique *	%	0.046	0.037	0.026	0.051	0.026	0.032	0.054	0.049	0.017	0.131	0.056	0.032	0.050
Carbone organique	%	0.98	1.05	1.19	1.12	0.7	1.04	1.4	1.43	0.86	3.06	1.25	0.86	1.25
Matière organique	%	1.69	1.82	2.05	1.92	1.21	1.79	2.42	2.46	1.48	5.28	2.16	1.48	2.16
Capacité Echange Cationique	meq.100g^{-1}	5	5	7	7	3	6	6	5	4	16	4	4	6
Calcaire Total *	%	0.18	0.16	0.34	0.46	0.34	0.18	0.57	0.34	0.34	0.57	0.23	0.11	0.35
pH *	-	7	6	6	7	7	5	6	5	6	5	7	7	6
Paramètres calculés* (Brisson et al., 2008)														
q0	mm	6.28	5.41	7.21	7.48	5.94	5.49	7.21	6.28	5.30	16.66	5.60	5.60	7.50
Albedo	-	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.28	0.25	0.25	0.25
Inflitration														
Horizon 1	-	50.0	50.0	12.7	12.7	50.0	50.0	12.7	50.0	50.0	0.9	50.0	50.0	38.4
Horizon 2	-	34.5	34.5	6.9	6.9	34.5	34.5	6.9	34.5	34.5	0.3	34.5	34.5	26.2
Horizon 3	-	18.3	18.3	3.5	3.5	18.3	18.3	3.5	18.3	18.3	0.3	18.3	18.3	13.8

* Données utilisées pour paramétrer le modèle STICS

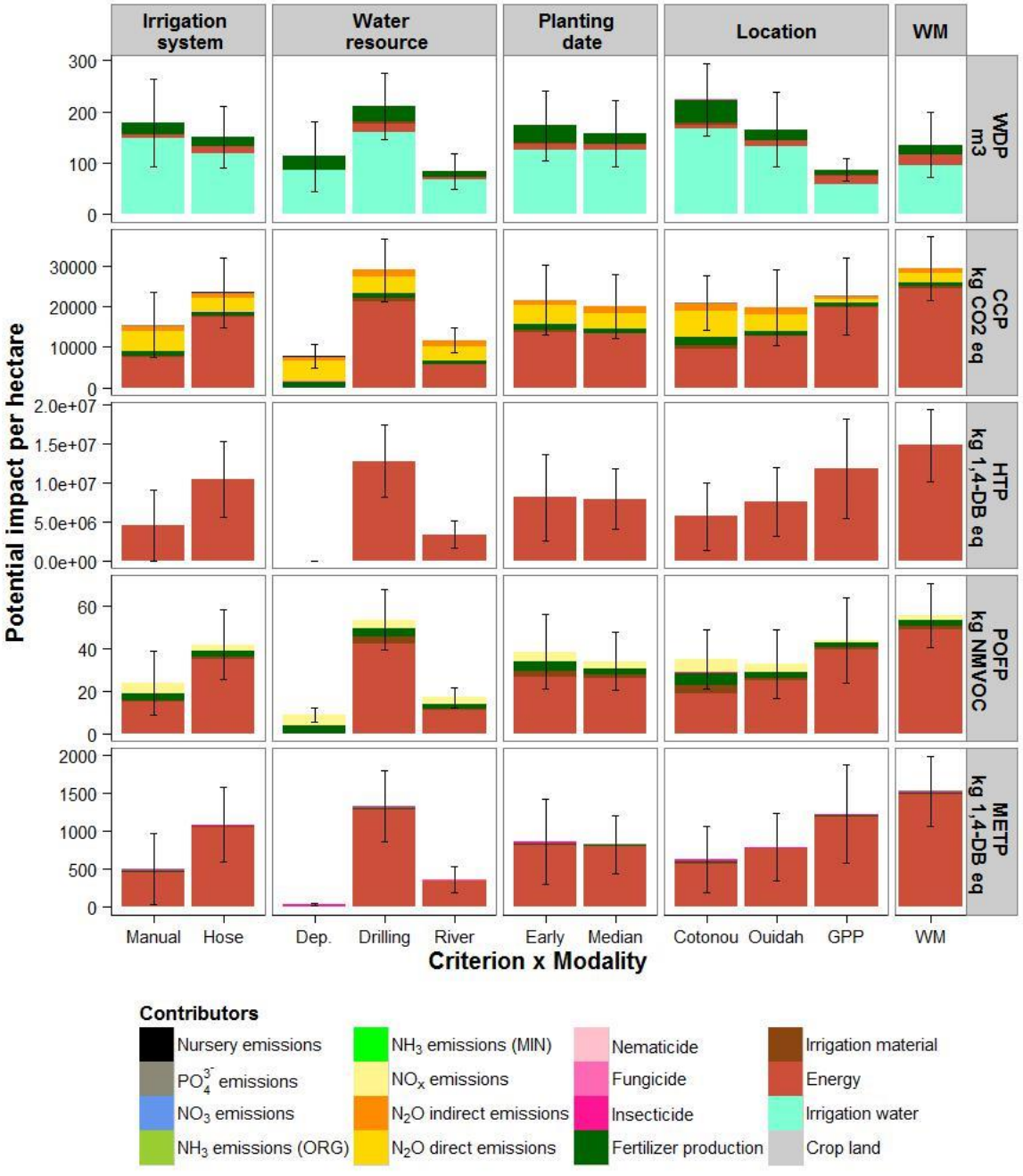
Annexe 7. Résumé des données climatiques produites par la station météo du centre de recherche AfricaRice localisé à Cotonou



Annex 8. LCIA results from cradle-to-harvest per hectare of out-of-season tomato in Benin for the 10 modalities accounting for 4 criteria, and the weighted mean

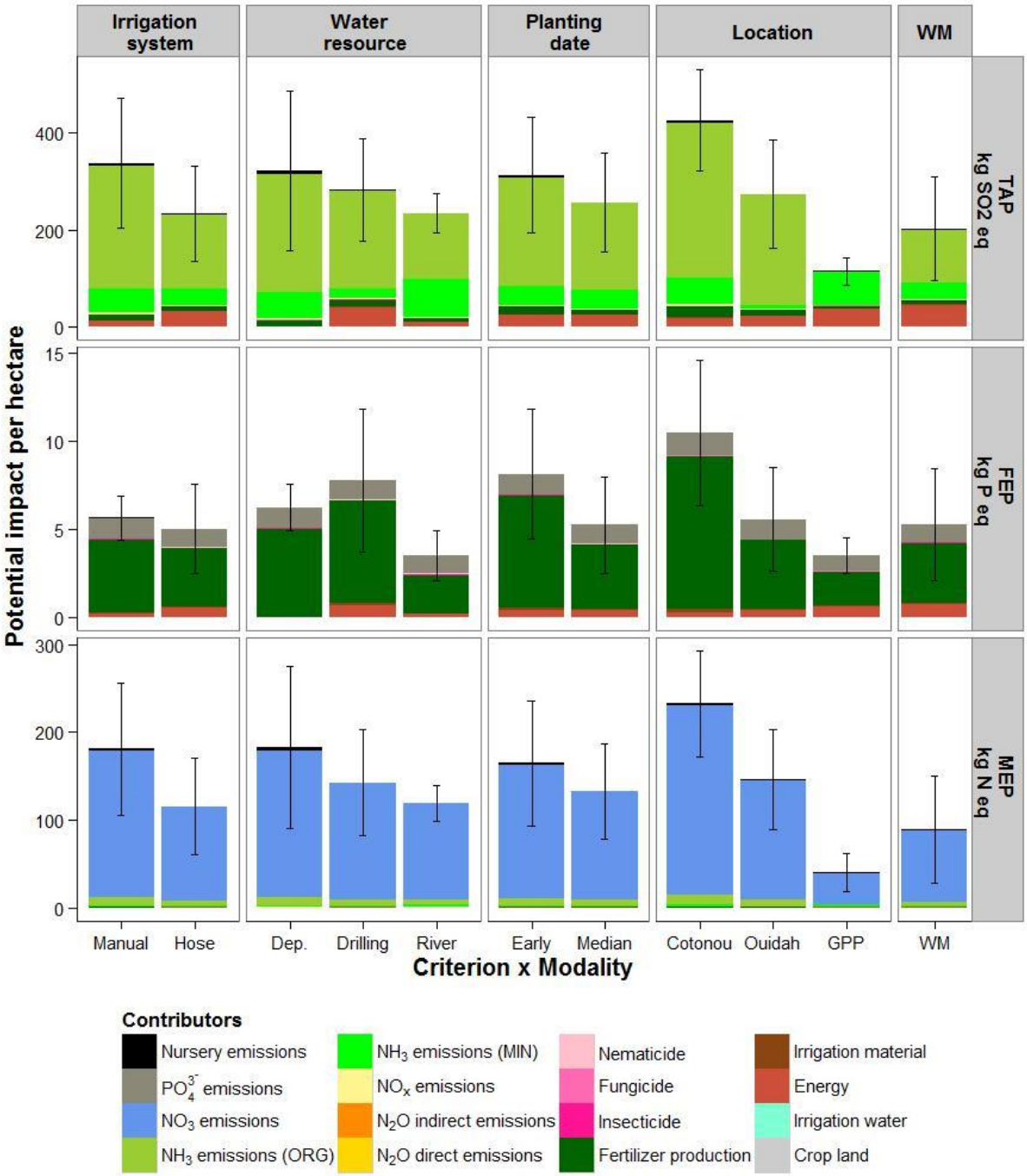
Impact categories influenced by irrigation practices:

- The water deprivation potential (WDP expressed in m^3 equivalent) calculated using water stress index (WSI) from Pfister et al. (2011),
- The climate change potential (CCP expressed in kg CO_2 equivalent),
- The human toxicity (HTP expressed in kg 1,4-DB equivalents)
- The photochemical ozone formation potential (POFP expressed in kg NMVOCs equivalent)
- Marine ecotoxicity potentials (METP expressed in kg 1,4-DB equivalents),



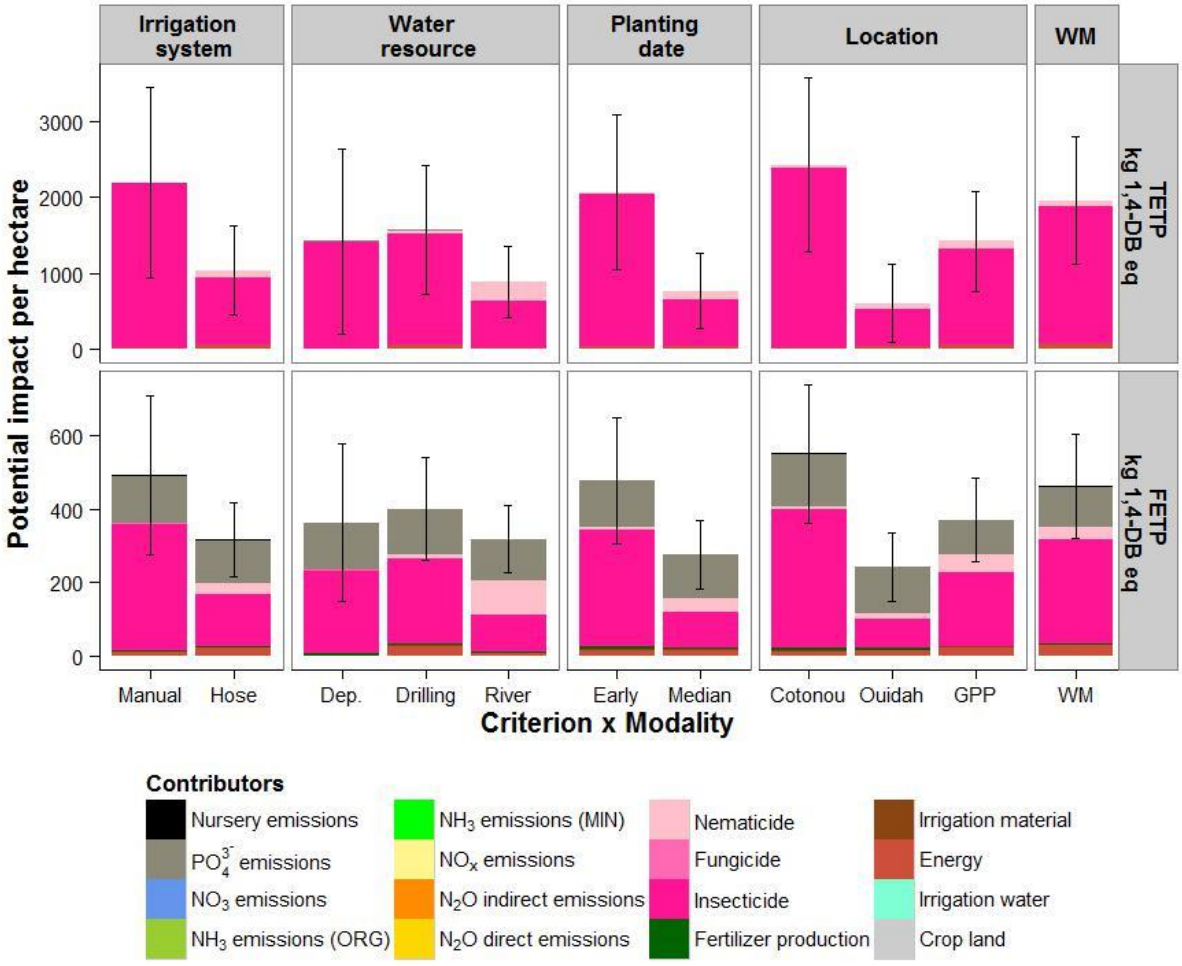
Impact categories influenced by fertilization practices:

- The terrestrial acidification potential (TAP, expressed in kg SO_2 equivalents)
- The freshwater eutrophication potential (FEP, expressed in kg P equivalent)
- The marine eutrophication potential (MEP, expressed in kg N equivalent)



Impact categories influenced by crop protection practices:

- Terrestrial ecotoxicity potentials (TETP expressed in kg 1,4-DB equivalents)
- Freshwater ecotoxicity potentials (FETP expressed in kg 1,4-DB equivalents),



Portefeuille de Compétences

Madame Aurélie PERRIN

Soutenance le 20/12/2013

Doctorat : Sciences agronomiques et écologiques

Ecole doctorale : ABIES - Agriculture, Alimentation, Biologie, Environnements et Santé

Etablissement : AgroParisTech

Encadrement : Benoît GABRIELLE, Mme Armelle BAEZA-SQUIBAN, Mme Claudine BASSET-MENS

Sujet de thèse : Quelle prise en compte de la variabilité sol, climat et pratiques dans l'analyse de cycle de vie des produits maraîchers ? Mise en œuvre au cas de la tomate

Ouverture scientifique et culturelle

- Journées ABIES 29 et 30 Mars 2011 - 29 mars 2011 - *14 heures. Mis en place par : ABIES - Agriculture, Alimentation, Biologie, Environnements et Santé*
- Journées ABIES 28 et 29 février 2012 - 28 février 2012 - *14 heures. Mis en place par : ABIES - Agriculture, Alimentation, Biologie, Environnements et Santé*

Formations scientifiques

- Analyse environnementale du Cycle de Vie - - *21 heures*

Formations à la recherche

- Communication scientifique en Anglais et Français - *28 heures*

Formations Langues Étrangères et F.L.E

- Atelier d'écriture d'articles scientifiques en anglais - *28 heures*

Formation aux logiciels

- EndNote - *14 heures*

Cours scientifiques et techniques

- Analyse de sensibilité, propagation d'incertitudes et exploration numérique de modèles en sciences de l'environnement - *27 heures*
- ACV: approfondissement du cadre conceptuel (14 au 16 mai 2012) - *15 heures*

Conférences

- 12th congress of the European Society of Agronomie - - *20 heures*
- 8th conference on Life Cycle Assessment in the Agri-Food Sector - - *22 heures*

Connaissance du Milieu Professionnel

- Préparation à l'insertion professionnelle - Objectif premier emploi - Public : doctorants en 3è année de thèse - Novembre 2012 - *24 heures. Mis en place par : ABIES - Agriculture, Alimentation, Biologie, Environnements et Santé*

Productions scientifiques

- **Aurélie Perrin, Claudine Basset-Mens and Benoît Gabrielle**. Life cycle assessment of vegetable products: a review focusing on cropping systems diversity and the estimation of field emissions *The International Journal of Life Cycle Assessment*

Autres valorisations techniques

- Farm and crop models:
  - × APSIM (Agricultural Production Systems sIMulator)
  - × APES (Agricultural Production and Externalities Simulator) STICS(Simulateur mulTIdiscplinaire pour les Cultures Standard)

- Computer: Word, Excel, Power Point, Access and Internet & notions about SPSS, R, SQL(PgAdmin) and Java(Eclipse)
- Agronomie, Analyse du cycle de vie et évaluation environnementale, Typologie, Flux d'azote à l'échelle de la parcelle

Enseignements et encadrement de stages

- Cours à Montpellier Supagro -Introduction à l'analyse du cycle de vie appliquée aux systèmes horticoles tropicaux- 2x 1.5h Ecole Chercheur ELSA- Introduction à la Typologie, usage en ACV- 1.5h
- Wilfried Yehouessi- Etude des performances agronomiques de la culture de tomate de contre saison dans la zone côtière du Sud – Bénin (Cotonou – Pahou – Grand-Popo). Octobre 2011- 6 mois. Université Abomey Calavi Benin, stage Master 1.
- Thais Juliane Do Prado : Prise en compte des techniques d'application et du l'état de la couverture végétale dans l'estimation des émissions de pesticides vers l'environnement. Juin 2012. 2 mois. Université de l'Etat de Sao Polo, Bresil, stage Master 1.

CV - Aurélie Perrin



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Née le 31/12/1985 à Dijon

OBJECTIF Mettre à contribution mes compétences pour le développement des systèmes de culture dans un objectif de réduire la dépendance aux intrants et plus particulièrement aux fertilisants.

EXPERIENCE

Doctorante | **UR HORTSYS- CIRAD**- Montpellier
20 Septembre 2010 – Décembre 2013

Gérer un projet de 3 ans, créer des données sur les systèmes de production de tomate au Bénin, rédiger des publications scientifiques et des rapports techniques pour les partenaires, utiliser des modèles de culture existants en collaboration avec les développeurs, superviser des stagiaires M1 et M2, présenter des résultats en anglais dans des conférences internationales.

Chargé de mission | **AIVB-LR** (interprofession des viticulteurs bio)- Montpellier
1er Avril 2010 – 27 Aout 2010

Collecter les informations sur les engrais et amendements organiques bio auprès des industries, caractérisation ces produits et de leurs usages pour une meilleure gestion de la fertilisation.

Chargé de mission | UMR SYSTEM- Montpellier Supagro
1er Novembre 2009 – 05 Septembre 2010

Coordonner la communication au sein du comité scientifique pour la mise en place des programmes et du contenue des conférences, créer et gérer la plate-forme web dédiée aux participants du congrès.

Stage de fin d'étude | PPS group-Wageningen University-Pay-Bas
15 Mars 2009 – 15 Septembre 2009

Evaluer de manière ex-ante les systèmes de cultures avec un focus sur les pratiques de fertilisation azotées, calibrer et évaluer un outil de modélisation (Blé, Pays-Bas).

Stage en cours de formation | APSRU-CSIRO-Toowoomba, Australie
01 Juin 2008 – 15 Septembre 2008

Etudier la production fourragère en milieu sub-tropical à tendance aride, conduire des expérimentations pour la collecte de données physiologiques et pédologiques.

DIPLOMES

Montpellier Supagro (34)
2009 Diplôme d'ingénieur Agronome spécialité Production Végétale Durable

LEGTA Etienne Munier (70)
2005 BTSA Technologies végétales option Agronomie et système de culture

Lycée Ste Bénigne (21)
2003 Baccalauréat scientifique option SVT

AUTRES COMPETENCES

- **Langues:** Anglais courant parlé et écrit (niveau C1), Espagnol parlé et écrit (niveau A2)
- **Informatique :** Pack Office, SQL (PgAdmin®), Java (Eclipse®), Html (eZpublish®), R (statistique), Simapro® (ACV)

PUBLICATIONS SCIENTIFIQUES

Conférences internationales

Perrin A, Basset-Mens C, Yehouessi W, Huat J, Gabrielle B, and Malezieux E (2012) Agronomical analysis of Life-Cycle Assessment (LCA) variability for different horticultural cropping systems at regional scale. *In the 12th Congress of the European Society for Agronomy.* August 20-24, Helsinki, Finland.

Perrin A, Basset-Mens C, Yehouessi W, Huat J, and Gabrielle B (2012) An LCIA-based typology for more representative results and refined data collection of a horticultural cropping system in the Tropics. *In the 8th International Conference on LCA of the Agri-Food sector (LCA Food).* October 1-4, Saint-Malo, France.

Perrin A, Basset-Mens C, and Gabrielle B (2013) The use of a mechanistic model to better estimate Nitrogen reactive (Nr) emission for Life-Cycle Assessment (LCA) of Tomato production in Benin, West Africa. *Under review for the 8th Australian Conference on Life Cycle Assessment (ALCAS 2013).* July 15-18, Sydney, Australia.

Burns M, **Perrin A**, Roux P, and Basset-Mens C (2013) Do the current life cycle inventory (LCI) methods adequately account for the life of a pesticide in horticultural production systems? A review. *Under review for the 8th Australian Conference on Life Cycle Assessment (ALCAS 2013).* July 15-18, Sydney, Australia.

Journaux à comité de lecture

Adam M, Belhouchette H, Corbeels M, Ewert F, **Perrin A**, Casellas E, Celette F, and Wery J (2012) Protocol to support model selection and evaluation in a modular crop modelling framework: An application for simulating crop response to nitrogen supply. *Computers and Electronics in Agriculture* **86**, 43-54.

Perrin A, Basset-Mens C, and Gabrielle B (2013) Life cycle assessment of horticultural cropping systems: a review focusing on systems variability and the estimation of field emissions. *The Internationnal Journal of Life Cycle Assessment DOI 10.1007/s11367-014-0724-3*

Perrin A, Yehouessi W, Huat J, and Basset-Mens C (2013) Methods and data for producing representative LCA results in a context of extreme data scarcity – the case of out-of-season tomato from Benin, West Africa. *Under review*

CENTRES D'INTÉRÊTS

Voyages pour découvrir d'autres cultures et paysages, Sports et loisirs : équitation, escalade, randonnée, ski de randonnée.

RESUME : L'agriculture urbaine représente une opportunité de réduire la pauvreté et d'améliorer la sécurité alimentaire des habitants des villes d'Afrique de l'Ouest. L'objectif général de cette thèse est de produire des données d'inventaire représentatives ainsi qu'une évaluation environnementale robuste de ces systèmes de production par la méthodologie Analyse du Cycle de Vie (ACV). Notre cas d'étude a été la tomate des jardins urbains au Bénin. Notre état de l'art nous a permis d'identifier que prendre en compte la diversité des systèmes de production et la variabilité des émissions au champ étaient deux enjeux majeurs de l'ACV des produits maraîchers. Nous avons donc développé un protocole de collecte de données basé sur la typologie qui prend en compte la diversité des systèmes puis mis au point une méthode d'estimation des flux d'azote au champ combinant un bilan d'azote et l'usage d'un modèle biophysique. Nous avons ainsi créé des inventaires pour 6 types de systèmes et une moyenne pondérée représentative de la population étudiée. L'analyse des performances agronomiques de ces systèmes a montré une forte variabilité des rendements et une utilisation fluctuante et souvent excessive d'engrais et de pesticides. L'exploration de la variabilité des flux d'azote à l'échelle de la parcelle et du cycle de culture a permis l'identification de 4 facteurs majeurs: les volumes d'eau d'irrigation, la dose d'azote, le pH du sol et la capacité au champ. A l'aide de scénarios favorables et défavorables aux émissions pour chacun de ces 4 facteurs, nous avons montré que leurs effets sur les résultats d'ACV étaient importants. L'intégration de ces données dans une ACV finalisée a montré qu'un hectare de production de tomate au Bénin était plus impactant que les productions maraichères Européennes. Les avantages du climat favorable du sud Bénin à la production de tomate en contre saison sont annulés par les faibles performances des systèmes d'irrigation, l'usage fréquent d'insecticides et d'importantes émissions azotées. Des données mesurées et des connaissances nouvelles sont nécessaires sur ces systèmes pour valider et affiner nos conclusions.

MOTS-CLEFS : Inventaire du cycle de vie · Typologie · Méthodes d'estimation des émissions · Modèle biophysique · Analyse de sensibilité

ABSTRACT: Urban agriculture provides opportunities to reduce poverty and ensure food safety for cities inhabitants in West Africa. The general objective of this thesis is producing representative inventories and a robust environmental assessment for those production systems using the Life Cycle Assessment (LCA) methodology. Our case study was the tomato production in urban gardens in Benin. Our state of the art identified the integration of the diversity of systems and the variability of field emissions as two major challenges for the LCA of vegetable products. We therefore developed a typology-based protocol to collect cropping systems data that includes their diversity and an approach combining a nitrogen budget and the use of a biophysical model to estimate nitrogen field emissions. We created inventories for 6 cropping system types and one weighted mean representative for the urban tomato growers in Benin. The analysis of the agronomical performances of these systems highlighted the important yield variability and the variable and often excessive use of pesticides and fertilizers. The investigation of nitrogen fluxes variability at plot and crop cycle scales led to the identification of 4 major influencing factors: water use, nitrogen input, soil pH and field capacity. Using favorable and unfavorable scenarios for nitrogen emissions for each of these 4 factors, we demonstrated that the LCA results were sensitive to their variations. The implementation of LCA using those contrasted data showed that one hectare of tomato production in Benin was more impacting than European vegetable productions. The benefits from the favorable climate for producing out-of-season tomatoes were hampered by the low efficiency of irrigations systems, the frequent use of insecticides and large nitrogen emissions. Measured data and new knowledge on these systems are needed to validate and refine our conclusions.

KEYWORDS: Life-cycle inventory · Typology · Emissions estimation methods · Crop model · Sensitivity analysis



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