

**COMPARATIVE STUDY OF ORGANIC AND  
CONVENTIONAL VEGETABLE FARMING  
SYSTEMS USING MATERIAL AND  
SUBSTANCE FLOW ANALYSIS**

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## ABSTRACT

Increase in human population has led to increased demand for food production. Thus, it is essential to adopt sustainable agricultural development. Organic farming is generally seen as a sustainable agriculture practice with lower environmental impact however, the low yield is unable to meet the food demand. High production conventional agriculture is often associated with significant environmental impact. This has raised the debate on sustainability issues of organic and conventional production. This study investigates two organic (OF) and two conventional vegetable farms (CF) to assess the sustainability and efficiency of each farm with material and substance flow analysis (MFA and SFA) using STAN 2.5 software. Based on STAN model, the annual C balance were around  $6,315 \pm 2,529$ ,  $9,912 \pm 1,816$ ,  $-304 \pm 12,988$  and  $10,802 \pm 4,929$  kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2, OF1 and OF2, respectively. The C balance of CF1, CF2 and OF2 are classified as “sink” while OF1 stock change is classified as “source”. The C flow highlights that major C input was from photosynthesis that contributed about  $3,803 \pm 2,123$ ,  $358 \pm 40$ ,  $1,193 \pm 506$ ,  $3,944 \pm 3,107$  kg C ha<sup>-1</sup> y<sup>-1</sup> of C flux in CF1, CF2, OF1 and OF2, respectively. The N balance of all the study farms were categorized as N “sink” given the N balances of  $1,589 \pm 116$ ,  $1,605 \pm 8$ ,  $2,608 \pm 18$ , and  $912 \pm 220$  kg N ha<sup>-1</sup> year<sup>-1</sup> in CF1, CF2, OF1, and OF2, respectively. The primary N input at OF1 and OF2 was compost, which accounts for 81% ( $2,201$  kg N ha<sup>-1</sup> year<sup>-1</sup>) and 60% ( $815$  kg N ha<sup>-1</sup> year<sup>-1</sup>) of the total N input. Chemical fertilizer used at CF1 and CF2 were about  $1,334$  and  $941$  kg N ha<sup>-1</sup> y<sup>-1</sup> while chicken manure contributed  $343$  and  $572$  kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively. This study concludes farm level management plays an important role in achieving sustainable agriculture. The study demonstrates MFA and SFA with STAN software allow a comprehensive assessment of agri-indicators from different views and aspects.

## ABSTRAK

Peningkatan populasi manusia telah meningkatkan permintaan pengeluaran makanan. Ladang organik biasanya dilihat sebagai amalan pertanian yang mampan dan mempunyai kesan pencemaran alam sekitar yang lebih rendah tetapi hasilnya yang rendah tidak dapat memenuhi permintaan. Ladang konvensional yang mempunyai pengeluaran yang tinggi sering dikaitkan dengan pencemaran alam sekitar. Ini telah menimbulkan perdebatan mengenai isu-isu kemampanan pengeluaran ladang organik dan konvensional. Dalam kajian ini dua ladang sayur organik dan konvensional telah dinilai untuk mengetahui kemampanan dan kecekapan setiap ladang dengan analisis aliran bahan-bahan (MFA dan SFA) menggunakan perisian STAN 2.5. Berdasarkan model STAN, baki C tahunan adalah sekitar  $6,315 \pm 2,529$ ,  $9,912 \pm 1,816$ ,  $-304 \pm 12,988$  dan  $10,802 \pm 4,929$  kg C ha<sup>-1</sup> y<sup>-1</sup> di CF1, CF2, OF1 dan OF2, masing-masing. Baki C di CF1, CF2 dan OF2 dikelaskan sebagai “singki” manakala OF1 diklasifikasikan sebagai “sumber”. Aliran C membuktikan bahawa kemasukan C utama adalah dari fotosintesis yang menyumbang kira-kira  $3,803 \pm 2,123$ ,  $358 \pm 40$ ,  $1,193 \pm 506$ ,  $3,944 \pm 3,107$  kg C ha<sup>-1</sup> y<sup>-1</sup> di CF1, CF2, OF1 dan OF2, masing-masing. Baki N dari semua ladang kajian dikategorikan sebagai N “singki” dengan  $1,589 \pm 116$ ,  $1,605 \pm 8$ ,  $2,608 \pm 18$ , dan  $912 \pm 220$  kg N ha<sup>-1</sup> y<sup>-1</sup> baki N di CF1, CF2, OF1 dan OF2, masing-masing. Kemasukan N utama di OF1 dan OF2 adalah melalui kompos yang menyumbang 81% ( $2,201$  kg N ha<sup>-1</sup> y<sup>-1</sup>) dan 60% ( $815$  kg N ha<sup>-1</sup> y<sup>-1</sup>) daripada jumlah N. Baja kimia yang digunakan di CF1 dan CF2 menyumbang kira-kira  $1,334$  dan  $941$  kg N ha<sup>-1</sup> y<sup>-1</sup> manakala tinja ayam menyumbang  $343$  dan  $572$  kg N ha<sup>-1</sup> y<sup>-1</sup>, masing-masing. Kajian ini menunjukkan bahawa pengurusan ladang memainkan peranan yang penting dalam mencapai pertanian lestari. Kajian ini menunjukkan MFA dan SFA dengan perisian STAN membolehkan penilaian yang komprehensif di buat bagi ladang dari pelbagai aspek dan segi.

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## LIST OF SYMBOLS AND ABBREVIATIONS

ASEAN	The Association of Southeast Asian Nations
ASOA	ASEAN Standard for Organic Agriculture
BMP	Best Management Practices
BNF	Biological N fixation
BOD	biochemical oxygen
C	Carbon
°C	Degree Celsius
Ca	Calcium
Cd	Cadmium
CEC	cation exchange capacity
CETDEM	Center for Environment, Technology and Development
CF	Conventional farm
CF1	Conventional farm number 1
CF2	Conventional farm number 2
CFU	Colony form unit
CH <sub>4</sub>	Methane
CHNS	Carbon, Hydrogen, Nitrogen and Sulfur
cm	centimeter
C:N	Carbon and nitrogen ratio
CO	Carbon monoxide
CO <sub>2</sub>	Carbon dioxide
CO <sub>2</sub> E	Carbon dioxide equivalent
COD	chemical oxygen demand
CVP	Cost volume profit

d <sup>-1</sup>	per day
DEFRA	UK Department for Environment, Food & Rural Affairs
DOA	Department of Agriculture Malaysia
DPN3	The 3rd National Agriculture Policy
<i>E. coli</i>	<i>Escherichia coli</i>
EC	<i>Escherichia coli</i> and coliform
EC	European countries
EEA	European Economic Area
e.g.	for example
EU	European Union
EU-27	European Commission
Eurostat	European Union Statistical Office
FAO	Food and Agriculture Organization of the United Nations
FDA	Food and Drug Administration
FSMA	Food Safety Modernization Act
g	gram
GDP	Gross Domestic Product
GHG	Greenhouse gas
GPP	gross primary productivity
Gt	Giga tonne
Ha	Hectare
HACCP	Hazard Analysis and Critical Control Point
IFOAM	International Federation of Organic Agriculture Movements
ICP-MS	Inductively coupled plasma mass spectrometry
IPCC	Intergovernmental Panel on Climate Change
IPM	Integrated Pest Management

IRENA	Indicator Reporting on the Integration of Environmental Concerns into Agriculture Policy
JAS	Japanese Agricultural Standard
K°	degree Kelvin
kg	Kilogram
km <sup>2</sup>	kilo meter square
Kt	Kilo tonnes
L	litre
MFA	Material flow analysis
m	meter
m <sup>3</sup>	cubic meter
Mg	Magnesium
mg	milligram
min	minute
mm	milimeter
Mn	Manganese
mth	month
MyGAP	Malaysian Certification Scheme for Good Agricultural Practice
N	Nitrogen
NA	Not available
NAP3	Third National Agriculture Policy
NASAA	National Association for Sustainable Agriculture from Australia,
ND	Not detectable
NH <sub>3</sub>	Ammonia
NH <sub>4</sub>	Ammonium
NO <sub>2</sub>	Nitrite

NO <sub>3</sub>	Nitrate
N <sub>2</sub> O	Nitrous oxide
NOP	The National Organic Program from United States
NPS	Non-point source
NUE	Nitrogen use efficiency
NUEP	Nitrogen use efficiency of productivity
OECD-FAO	Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations
OF	Organic farm
OF1	Organic farm number 1
OF2	Organic farm number 2
ÖNORM	National standard published by the Austrian Standards Institute
P	Phosphorus
RMK-9	Ninth Malaysia Plan “Rancangan Malaysia ke 9”
RMK-10	Tenth Malaysia Plan “Rancangan Malaysia ke 10”
RMK-11	Eleven Malaysia Plan “Rancangan Malaysia ke 11”
Pg	Picogram
Pb	Lead
PO <sub>4</sub>	Phosphate
ppm	part per million
RM	Ringgit Malaysia
SALM	Good Agricultural Practices “Skim Amalan Ladang Baik”
SALT	Livestock Farm Practices Scheme
SFA	Substance flow analysis
SOC	Soil organic carbon
SOM	Malaysian Organic Scheme Certification

SOM	Soil organic matter
SPLAM	Malaysian Aquaculture Farm Certification Scheme
SS	<i>Salmonella sp.</i> and <i>Shigella sp.</i>
STAN	subSTance flow ANalysis
t	tonne
Tg	Teragram
TKPM	Permanent Food Park Programme “Taman Kekal Pengeluaran Makanan”
TSS	total suspended solid
UK	United Kingdom
µL	micro litre
UNESCAP	United Nations Economic and Social Commission for Asia and the Pacific
UNFCC	United Nations Framework Convention on Climate Change
US	United States
USD	United States Dollar
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
WHO	World Health Organization
WUEP	Water use efficiency of productivity
y	year
3R	Reduce, Reuse, Recycle



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University of Malaya

## CHAPTER 1: INTRODUCTION

### 1.1 Introduction

Over 2.3 billion world population is expected between 2009 and 2050 (FAO, 2009; Gerland *et al.*, 2014; Soubbotina, 2004). In 2013, the average world population growth rate was recorded as 1.3 % and Malaysia's growth rate was recorded as 1.6% (MPC, 2014). This means that food demand will continue to rise with the projection of 3 billion tonnes of cereals, for both food and animal feed (FAO, 2009). Even though, the global population is forecasted to be stabilized in recent years, but the present large population base still require an increase in food supply. The OECD-FAO forecasted that over 60% of agricultural production increments are required within the next 40 years, in order to meet rising food demand (OECD & FAO, 2012). Developing countries such as Malaysia would need to double the food production in order to accommodate the growth (FAO, 2009). According to Malaysian Department of Agriculture, in 2011, the average vegetable consumption in Malaysia was 55.3 kg per capita per year and the total vegetable production was 928,183 metric tons (DOA, 2012). However, such vegetable production capacity was unable to meet Malaysia's demand and the country is still highly depended on imported vegetables. In 2011, a total of RM 2,734,600 ( $\approx$  USD 638,563) of vegetables was imported by Malaysia (DOA, 2012). The 10th Malaysia Plan has been formulated with the Agri-Food Policy in mind, with the objectives to ensure adequate food supply for the country and thus leads to recent increased number of vegetable farms.

Intensive agriculture activities often associate with significant environmental impact such as deforestation, pollution, greenhouse gas emission, soil quality change and reduction of biological activity which has compromised food production, environment

and social safety (FAO, 2015c; Stoate *et al.*, 2001). In addition, the competition for water and land are also the major concerns of intensified agriculture production (Willenbockel, 2014). Intensive agriculture is one of the biggest causes of diffuse water pollution globally due to nutrients from fertiliser and manure, silt from soil erosion, pesticides and herbicide (Hooda *et al.*, 2000; Nie *et al.*, 2012). It is also the primary and largest source of N pollution. Globally each year, around 140 million tonnes of N is lost to the environment as ammonia, N oxides and other compounds (Qiu, 2013). By 2050, it is estimated that the world fertilizer consumption would increase by 50% and the global N loss to environment would increase by 70% (Sutton & Bleeker, 2013). Various problems are associated with N pollution such as soil acidification, harmful algal blooms and threatening biodiversity. The largest impacts of N pollution would be on freshwater and marine ecosystems, which would be greatly eutrophied by high rates of N release from agricultural fields (Bouwman *et al.*, 2013). Thus, efficient use of added N is highly important in all arable farming systems, as the transport of N in runoff and drainage from agriculture soils causes pollution of surface waters (Stenberg *et al.*, 2012). Agricultural intensification also would contribute to atmospheric accumulation of greenhouse gases (Cline, 2007). About 13.5% of global GHG emissions are due to crop and livestock production (FAO, 2014b). Almost half of the global greenhouse gases is contributed by N<sub>2</sub>O emission from agricultural systems (Yang *et al.*, 2014a). Agriculture has significant effects on climate change, primarily through land use change and greenhouse gases emissions (Willenbockel, 2014). Biomass burning, crop production and conversion of grasslands to croplands are the primary human activities that increase atmospheric CO<sub>2</sub>. Biomass burning can contribute atmospheric emission of trace gases CO, H<sub>2</sub>, CH<sub>4</sub>, N<sub>2</sub>O, NO<sub>x</sub> (NO and NO<sub>2</sub>), COS and CH<sub>3</sub>Cl.(Crutzen *et al.*, 2016)

The key limiting factors of crop growth are water, mineral nitrogen, and mineral phosphate and these are often supplied in excess by farmer to maximize crop yields (Rosen & Eliason, 1996). Improper farm practices are the main reason of environmental pollution. Practices such as over tillage, excessive synthetic input (herbicide and pesticide), excessive fertilization, and intensive irrigation can result in detrimental environmental impacts (Killebrew & Wolff, 2010). The practice of improper farm management often increased the risk of environmental pollution (Gibbons *et al.*, 2014; Tilman, 1999). Environment impacts in agriculture can be minimized only if there are much more efficient and sustainable farm management (Goulding *et al.*, 2008; Horrigan *et al.*, 2002; Lal, 1993, 2009; SARE, 2010; Wezel *et al.*, 2014; Wu & Ma, 2015). The improvement of agricultural system comes from better innovation, technology, regulatory and agronomic practices with commercial strategy. Adaptive farm management plays an important role in sustainable development of the agricultural sector.

Proper land use management could reduce N loss to the environment, improve resource use, mitigate GHG emission or even create C sink by encouraging C sequestration practices (Freibauer *et al.*, 2004; Luo *et al.*, 2010; Ogle *et al.*, 2005; Vleeshouwers & Verhagen, 2002). Terrestrial ecosystem is an ideal reservoir for carbon sequestration and could offset the CO<sub>2</sub> emission due to human activities (Luo *et al.*, 2010). The IPCC has identified biomass input to soil as the promising tool to capture and store C at terrestrial reservoir (Sims *et al.*, 2007). Farm C input is the key factor of soil organic matter turnover rates which are known to exert high influence over soil carbon content (Freibauer *et al.*, 2004). The management of C and N flow plays a crucial role in environment, climate and human health protection while ensuring sufficient food production (Goulding *et al.*, 2008; Wezel *et al.*, 2014).

Vegetables are one of the cheapest sources of nutrient, minerals, antioxidant and vitamins which play an important role in human health especially in developing countries with high population growth (Khan *et al.*, 2008; Schreiner, 2005). However, the typical vegetable productions required high fertilizer application with frequent irrigation due to the shallow roots that limits the water and nutrient up take efficiencies which poses a greater environmental challenge as compare to other crop types (Hartz, 2006). Therefore, this study selected vegetable farms as the main study sites.

## **1.2 Sustainable agriculture**

Increased crop productions and productivity cannot come at the expense of the environment (Campanelli & Canali, 2012; Chen, 2011). Thus, it is necessary to adopt sustainable agricultural development such as resource conservation, environmental impact mitigation, global climate change mitigation and adaptation, which are integral to any agricultural program that aimed to increase production. A sustainable farm system should ensure balance between environments, economic and social by ensuring efficient production while conserving resources and the environment (Duesterhaus, 1990).

Organic farming is generally seen as an innovative and sustainable agriculture practice that can have a lower impact on the environment (Cestti *et al.*, 2003; Hartz, 2006; Lichtfouse *et al.*, 2009; Wezel *et al.*, 2014). Organic farming is an approach with inter- and trans-disciplinary sciences and concept of naturalness (IAASTD, 2009; IFOAM, 2015a). Organic farming is considered the driving force to ensure the country's economic growth, environmental protection, food security and public health (Hansen *et al.*, 2002). Based on Rodale Institute's 30 years of soil carbon data, organic farming is a

very potential carbon sequester and is consider as one of the important solution for global warming (LaSalle & Hepperly, 2008). It is believed that the common practices of organic material application resulted in C sequestration (Paustian *et al.*, 1992). Various reports tried to conclude the benefit of converting from conventional farm to organic farm in regards to C sequestration (Eve *et al.*, 2002; Gattinger *et al.*, 2012; Liu *et al.*, 2013b). Evidence of higher soil C concentration in organically managed farm was found, yet some other studies have not agreed with such findings (Janzen, 2006; Leifeld & Fuhrer, 2010; Scialabba & Müller-Lindenlauf, 2010). Several modelling studies reveal that conversion of conventiona farm to organic farm increases soil C is only a temporary solution for C sequestration (Foereid & Høgh-Jensen, 2004; Smith, 2004). In addition, there is also potential risk of nutrient loss from the organic system due to excessive nutrient enrichment by repeated application of manure and compost (Hartz, 2006; Wyland *et al.*, 1996).

In life cycle assessment point of view, organic farms may not be as environmental friendly as they are thought of. According to Venkat (2012), seven out twelve case studies for conventional farms transformed into organic farms indicated that organic farms released higher amount of greenhouse gas as compared to conventional farms. This is supported by Nguyen *et al.* (1995) and Korsæth (2008), where conventional farm production rate ranged from 16% to 71% higher than organic farm which means that higher land used is needed for organic farm. However, higher yield in conventional farm often associated with such improper farm practices such as over tillage, excessive use of fertilizer and pesticide and exploitation of soil (Killebrew & Wolff, 2010).

This has again raised the sustainability issues between organic farm and conventional farm. The increase in population leads to increasing demand for land and food, thus the

ideal cropping system should maximize food production per unit area and at the same time minimize the undesirable environmental effect (Korsaeth, 2008). This highlights the importance of proper farm management regardless of the farm types. The present study aims to analyse variation in C and N mass flow between fields representing organic and conventional farm systems with MFA/SFA as an alternative assessment tool.

### **1.3 Material/substance flow analysis**

Material/substance flow analysis (MFA/SFA) has been widely used to trace the flow of production, use, and consumption of materials or element for various economic sectors and discipline, e.g. industrial management, industrial ecology, waste management, architecture, ecological, energy, environment and agriculture (Bailey *et al.*, 2004; Davis *et al.*, 2007; Fuse & Tsunemi, 2012; Guo *et al.*, 2015; Hashimoto *et al.*, 2007; Hawkins *et al.*, 2007; Huang *et al.*, 2012; Nakajima *et al.*, 2013; Nakamura & Nakajima, 2005; Sendra *et al.*, 2007; Smit *et al.*, 2015). It is a decision tool used for early recognition, priority setting, to analyse and improve the effectiveness of measures and to design efficient resource management strategies in view of sustainability (Hendriks *et al.*, 2000; Huang *et al.*, 2012). It is also a practical analytical method to quantify flows and stocks of materials or substances in a defined spatial and system (Baccini & Brunner, 2012; Brunner & Rechberger, 2004). The MFA and SFA has become a useful tool as environment indicator, eco-efficiency indicator and industrial ecology (Sendra *et al.*, 2007; Wang *et al.*, 2016).

A farm system has a unique characteristic and metabolism. The metabolism of a farm system can be managed to achieve sustainable resource management and environmental development (Jakrawatana *et al.*, 2015). The advantages in utilizing MFA/SFA for farm

level analysis are its ability to quantify all material flows and identifying significant and simple indicators that can discover critical points and demonstrate the metabolism state and changes of the farm system (Sendra *et al.*, 2007). Therefore, MFA and SFA provides information which plays a vital role in determining the farm system stability by focusing on tracing mass and substances flow within the farm system boundaries. The model emphasized the imminent resource and environmental issue without depending on indicators of environmental stress (Hendriks *et al.*, 2000). MFA and SFA is a potential assessment tool to support the sustainable development of farms which function to:

- i. Build a systematic database or information pool to help formulate measures to improve the efficiency of farm management,
- ii. Determine critical links or pathways of losses for agri-environment monitoring,
- iii. Establish indicators bank by deriving meaningful and simple agri-environment indicators,
- iv. Optimizing resource use (Huang *et al.*, 2012).

With the current research, the MFA and SFA are applied as an alternative approach to farm systems monitoring. The MFA/SFA modelling is performed by using STAN 2.5 developed by Oliver Cencic of Vienna University of Technology. The software supports the performing of MFA/SFA according to the Austrian standard ÖNORM S 2096 (Material flow analysis - Application in waste management) under consideration of data uncertainties (Cencic & Rechberger, 2008; Laner *et al.*, 2014). The STAN 2.5 combines all necessary features of MFA in one software product: graphical modelling, data management, calculations and graphical presentation of the results. Further, STAN



model allows for structuring the SFA system according to key processes and stocks on several levels of processes and subordinate system (Vyzinkarova & Brunner, 2013).

The MFA and SFA principles integrated into farm management are able to optimize the resource use and minimize resource losses. In this study, the MFA/SFA is used to model C and N balance to evaluate organic and conventional leafy vegetable farm systems. By quantifying the C and N balance of the farm material cycle, various materials inputs like fertilizer, soil amendment and water and, outputs such as vegetable yield, evapotranspiration, gaseous emission, runoff and leaching are taken into account in order to optimize farm resource management. The study focused on six categories: soil, water, air, vegetable, resource use and economic. Below are the key indicators of environment impact assessment for this study (Table 1.1):

**Table 1.1: Key indicators of environment impact assessment**

<b>Category</b>	<b>Indicators</b>
Soil	C and N stock; pathogen level
Water	Water flow; ammonia, nitrite and nitrate content; pathogen level; N and C loss
Air	Ammonia, carbon dioxide, carbon monoxide, methane and nitrous oxide emission
Vegetable	Pathogen level; yield
Resource use	C and N balance; fertilizer use efficiency; water use efficiency; N and C use efficiency; waste generation
Economic	Cost-profit analysis

#### **1.4 Problem statements**

The establishment of Permanent Food Park Programme (TKPM) “Taman Kekal Pengeluaran Makanan” at every state in Malaysia has increased the land use for vegetable production. Agricultural expansion often links to agricultural pollution. Intensive vegetable crop production tend to supply large amounts of fertilizer and frequently irrigate the fields, leading to high risk of pollution (Ju *et al.*, 2006; Ju *et al.*,

2004; Song *et al.*, 2009). Malaysian Department of Agriculture has been promoting sustainable agriculture through certification such as Malaysian Certification Scheme for Good Agricultural Practice (MyGAP) and Malaysian Organic Scheme Certification (SOM). Results from farm level investigation in this study are necessary for the improvement of sustainability standards and certification and generate data of potential lossess, resource use, and agri-environment indicators.

Vegetable farm C and N flow assessment studies conducted by previous researches emphasized on temperate regions (Foereid & Høgh-Jensen, 2004; Ju *et al.*, 2006; Ogle *et al.*, 2005; Salo & Turtola, 2006; Song *et al.*, 2009). There are very few researches conducted in tropic region but mainly focusing on small scale vegetable farm (Abdulkadir *et al.*, 2013; Goenster *et al.*, 2014; Hedlund *et al.*, 2004; Hedlund *et al.*, 2003). On the other hand, carbon sequestration at tropics and sub-tropics region faced difficulties because of the high soil degradation rate (Lal, 2004b). The restoration of degraded soil and ecosystems in tropics and subtropics is much needed. The insufficient information of C storage in agriculture land was noticed especially in developing world, tropics and subtropics region (Govaerts *et al.*, 2009). Data limitation is the main set back in meta-analysis of global soil C change (Leifeld & Fuhrer, 2010). Therefore, presents study provides the vital information of C and N flux of vegetable farms in tropical regions.

Farm system is a dynamic system that is influenced by several factors ranging from natural conditions to farm management. This study distinguishes itself by demonstrating the C and N flow model of organic and conventional vegetable farm management in tropical area and a developing country. The farm level data offers a preview of farm C and N flow through case study at four existing farms in Malaysia. The farm balance was

modelled based on mass conservation theory. It highlights the existing and potential material stocks accumulating within a system which can cause environmental problems or serve as a potential source of resources.

## **1.5 Objectives**

The general objective of this study is to establish, model and analyse of the mass flow system of two organic and two conventional vegetable farms, in order to obtain proper approaches in improving the current farm management system. The main objectives of this study are as follows:

- i. To characterize conventional and organic vegetable farm
- ii. To generate mass balance of conventional and organic vegetable farm to identify potential C and N sink
- iii. To generate STAN model of material, C and N flow
- iv. To determine the greenhouse gas emissions from agriculture soil in evaluating global warming potential
- v. To determine the level of pathogenic contaminants in irrigation water, soils, and vegetables, and
- vi. To compare the performance of conventional and organic vegetable farms by evaluating the yield, nutrient use efficiency and cost profit analysis.

## CHAPTER 2: LITERATURE REVIEW

### 2.1 Introduction

This chapter provides a general background of agriculture in a sustainable perspective and describes the theories of material/substance flow analysis (MFA/SFA) and how it leads to the principles of integrated farm management. This will give alternative agri-environmental indicators and assessment tools to analyse, evaluate and model farm system for farm management improvement.

### 2.2 Global Agriculture

Over 2.3 billion world population growth is expected between 2009 and 2050 (Figure 2.1) (FAO, 2009; Gerland *et al.*, 2014; Soubbotina, 2004). Even though the global population is forecasted to be stable in recent years, the present large population base still requires an increase in food supplies. Figure 2.2 demonstrates the global changing trend of world population, cropping area, crop production and seasonal cycle of crop gross primary productivity (GPP) from 1961 until 2010. Within the last 50 years, the world population has doubled from 3 billion to 7 billion. In the same time, the cropping area and crop production showed increasing trend by about 20% and 300%, respectively (Zeng *et al.*, 2014).

Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations (OECD-FAO) Agriculture Outlook 2012-2021, forecasted that over 60% of agricultural production increments are required within the next 40 years, in order to meet this rising food demand (OECD & FAO, 2012). This implies that the food demand and agricultural production would continue to grow.



Figure 2.1: World population projection, 1950-2100 (Gerland *et al.*, 2014)

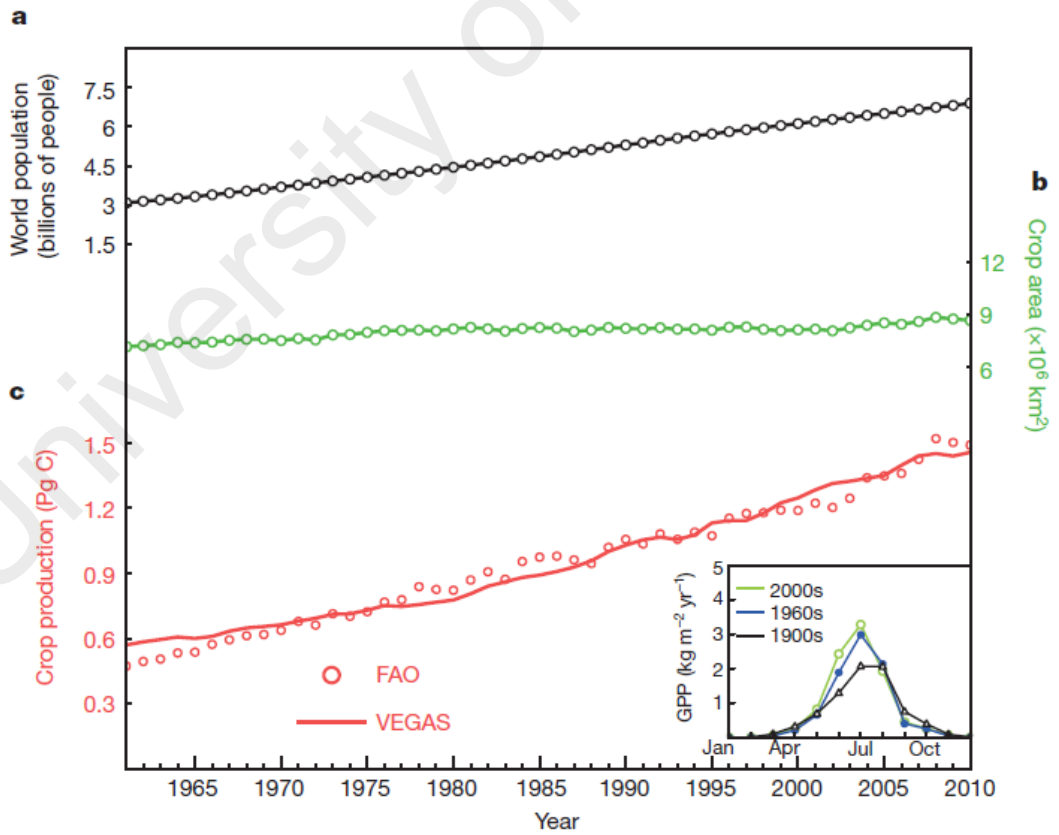


Figure 2.2: Annual world population (a), cropping area (b), crop production (c) and seasonal cycle of crop gross primary productivity (GPP) (Zeng *et al.*, 2014)

In addition, the rise of biofuel and bioenergy demands further intensified the need to increase agriculture production (Willenbockel, 2014). FAO (2009) projected that by 2050, the demand for cereals would reach 3 billion tonnes and the current demand for cereal is 2.1 billion tonnes. By 2050, agriculture production is required to increase by at least 70% to be able to meet the food demand and this could be by intensive agriculture or land expansion (Foley *et al.*, 2011; Godfray & Garnett, 2014). Production in developing countries such as Malaysia would need to be doubled (FAO, 2009).

### **2.3 Agriculture in Malaysia**

In the past 50 years, Malaysia has successfully undergone economic transformation from reliance on agriculture to industrial based. However, agriculture continues to play a crucial role in the overall economic growth of the country. The contribution of agriculture to the Gross Domestic Product (GDP) declined from 28.8% in 1970 to 7.3% in 2010. Although the contribution of agriculture to GDP showed a declining trend, the actual value of output and productivity has actually increased (DSM, 2012). Between 2011 and 2015, agricultural sector contributed RM455 billion ( $\approx$  USD 106 billion) to GDP with the average annual growth rate of 2.4 %. It is determined to maintain a momentum of 3.5 % growth, between 2016 and 2020 through modernization of the sector and strengthening of innovation and research and development by the support of the Eleven Malaysia Plan (RMK-11) (EPU, 2015).

Malaysia's agriculture has developed into a large scale, systematic, intensive, and market oriented modern agriculture. Malaysian agricultural sector is facing two major challenges:

- 1) National food security by ensuring sufficient food to meet the population growth and
- 2) Sufficient production of raw material for food manufacturing industry (Daud, 2004).

Developing countries such as Malaysia would need to double the production in order to accommodate the population growth (Anderson & Strutt, 2014; FAO, 2009). Industrialization of Malaysia has led to fast expansion of urban population that increased the conversion of agricultural and vegetation land for housing and township development (Samat *et al.*, 2014). The competition for land has pressured the agriculture sector for more efficient production with limited amount of land. In year 2011, there was a total of 4.9 million hectare of agriculture land which was about 15% of the total land mass of Malaysia (Table 2.1) (DOA, 2012).

Table 2.1: Agriculture land use in Malaysia (Ha) (DOA, 2012)

Category of land use	Type of land use	2010	2011
Farm and annual crop	Agricultural station	8,954	9,025
	Floriculture	2,376	2,945
	Gardening	325,376	327,329
	Vegetables	16,922	18,806
	Herbs and spices	282	1,546
Estate, plantation and perennial plants	Koko	12,988	8,739
	Coconut	104,490	95,708
	Rubber	1,277,352	1,311,947
	Coffee	1,539	863
	Oil palm	2,804,257	2,910,945
	Areca nut	674	244
	Sago	1,997	1,681
	Tea	2,411	2,399
Fruit farm	184,603	171,616	
Total		4,744,221	4,863,793

Figure 2.3 shows the agriculture field size (accuracy of >82.4%) in Malaysia generated with GeoWiki (Fritz *et al.*, 2015). The satellite image shows that majority of the agriculture fields are small and medium sized. This indicates the potential on agriculture land expansion. Based on the satellite image, a majority of the agriculture fields are mostly concentrated at the southern region and central region of Malaysia and therefore, this study has selected two farms from the southern region and two from central region.

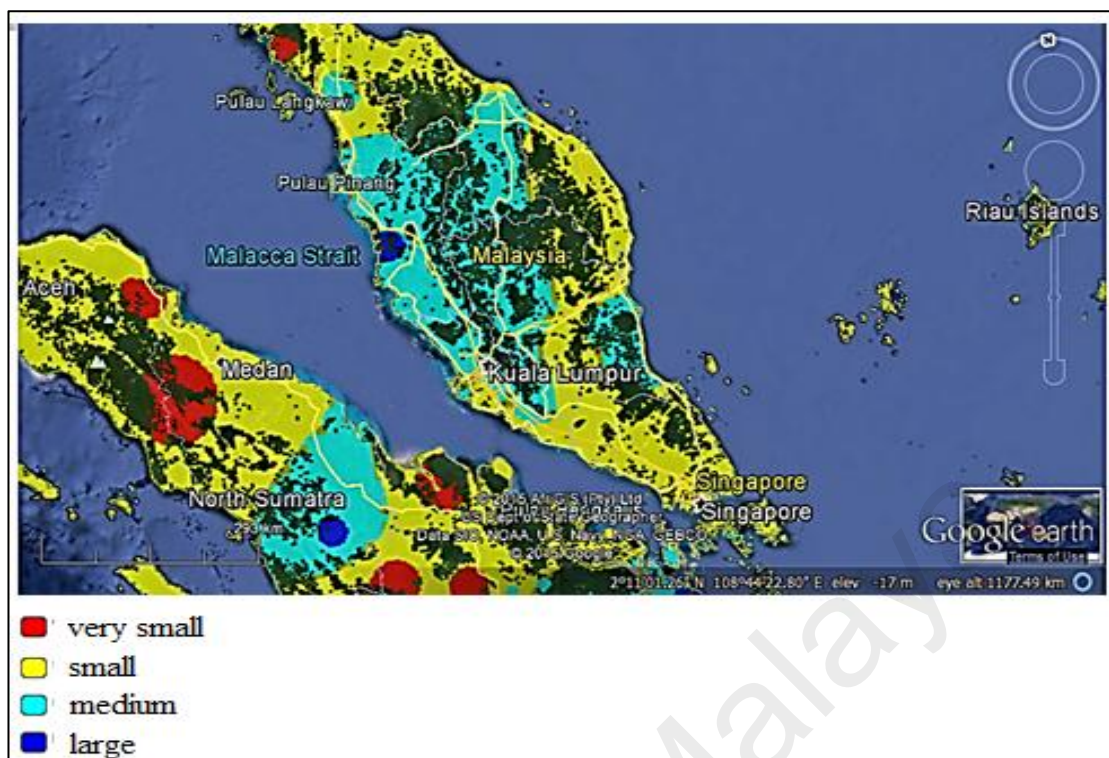


Figure 2.3: Satellite image showing agricultural field in Peninsular Malaysia, 2015.

Credit: IIASA; Geo-Wiki Project; Google View

High dependency on imported food indicates that Malaysian agricultural production has yet to reach the self-sufficient stage. Industrialisation has pushed a large scale of agriculture towards exporting (Indrani, 2001). However, Malaysia is yet to achieve a balance in import-export trading (Table 2.2). Instead of increasing cultivation, the agriculture production efficiency is the key for Malaysia to achieve self-sustainable agricultural production.

Table 2.2: Trade data - export, import and balance of trade for vegetables, Malaysia

(DOA, 2012)

Exports (Tonnes)	Imports (Tonnes)	Balance of Trade (Tonnes)
887,591	3,111,947	-2,224,356



The Permanent Food Park (TKPM) started with the 8<sup>th</sup> Malaysia Plan (RMK-8) and part under The 3rd National Agriculture Policy (DPN3) with the aim to promote the involvements of entrepreneurs and private sectors in large scale commercialised agriculture projects with the use of modern technology (DOA, 2014). Listed below are the objectives of TKPM projects:

- To develop a permanent zone for food production;
- To encourage the involvement of entrepreneurs and the private sector in a large scale and commercial farming.
- To increase the involvement of private sector as an anchor company to promote marketing and value added activity
- To target the net monthly income per participants of at least RM3000/month ( $\approx$ USD 845/month).
- To increase the national food production, high quality and sustainable and also improve the Good Agriculture Practice (GAP);

The TKPM project was continued under the 9<sup>th</sup> Malaysia Plan (RMK-9). Since then, a total of 800 participants with 4,339 ha of farm land have participated under this project and has successfully contributed to 110,000 tonnes of agriculture produce that valued at RM 113 million ( $\approx$ USD 32 million) (MOA, 2015). Government policy will continue to promote the agriculture sector and thus lead to increase in agriculture land in the country. The National Agriculture Policy and the 9<sup>th</sup> Malaysia Plan (2006-2010) are the recent efforts from the government to transform agriculture sector into a modern and competitive sector (Razak & Roff, 2007). The government aims to achieve this through increase in production efficiencies, optimal resource utilization, intensive land usage, as well as proper soil and water conservation. This effort also includes promoting organic and integrated farming.

## 2.4 Vegetable farming

Vegetables are one of the cheapest sources of nutrient, minerals, antioxidant and vitamins which play an important role in human health, especially in developing countries with high population growth (Khan *et al.*, 2008; Schreiner, 2005). Vegetable farming is defined as growing of vegetable crops, primarily for human consumption (Warid, 2015). Vegetable crops have high market demand and value. However, the vegetable productions pose an environmental challenge. This is because the typical vegetable productions require high fertilizer application with frequent irrigation due to the shallow roots that limits the efficiencies of water and nutrient use. In addition, the extensive tillage with no cover crops has a major environmental impact (Hartz, 2006).

Vegetable farming in Malaysia is a year-round cropping activity and often characterised as small, smallholder, scattered farm, close to urban areas, grown under open, greenhouse or netting and practised traditional farming methods (Arshad & Noh, 1994). There are a number of large farm (>25ha) cultivating under rain shelters located at the southern region of Malaysia which markets the produce to Singapore (Razak & Roff, 2007). In 2008, growing of vegetables had contributed to 0.34% of total gross output which is about RM 111.4 million ( $\approx$  USD 26 million) (DSM, 2012). Malaysia Department of Agriculture statistic reported that in 2011, the average vegetables consumption in Malaysia was 55.3 kg per capita per year and the total vegetables production was 928,183 metric tons (DOA, 2012). However, such vegetable production capacity was unable to meet Malaysia's demand and the country is still highly dependent on imported vegetables. In 2013, a total of RM 3,111,947 ( $\approx$ USD 876,802) vegetables were imported by Malaysia (DOA, 2012). The 10th Malaysia Plan (RMK-10) has formulated the Agri-Food Policy, with the objective to ensure self-sufficient food supply for the country and thus leads to recent increased number of vegetables farm. In

2013, in Malaysia there was a total of 67,777 ha vegetable farms with a production volume of 1,434,200 tonnes and money value of RM 4,773,989 ( $\approx$ USD 1,345,089) (Table 2.3). Between 2009 and 2013, there was an increase of almost 50% of total vegetable planting areas and the trend will continue to grow under government projects such as TKPM. Generally most of the vegetable farms in Malaysia are small (5-35 ha) and medium (35-225 ha) size which has the potential to expand under the promotion of government policies (Figure 2.4). Selangor and Johor region have the largest area of lowland vegetable farms after Pahang (highland vegetable cultivation) (Table 2.4). Therefore, the selected study farms are located at these two regions.

Table 2.3: Information on Malaysia vegetable cultivation, 2009-2013

	2009 <sup>a</sup>	2010 <sup>a</sup>	2011 <sup>a</sup>	2012 <sup>a</sup>	2013 <sup>b</sup>
<b>Planting area (Ha)</b>					
Malaysia	41,078	52,793	51,777	53,322	63,030
Peninsular Malaysia	34,487	45,378	43,654	45,833	54,946
Sabah	2,832	2,882	3,600	2,911	3,468
Sarawak	3,635	4,433	4,395	4,478	4,490
W.P Labuan	124	100	129	101	127
<b>Production (Tonnes)</b>					
Malaysia	623,457	871,630	928,183	878,975	1,326,504
Peninsular Malaysia	540,746	784,194	833,432	792,056	1,238,024
Sabah	38,061	39,346	47,771	39,741	42,050
Sarawak	43,262	45,613	45,959	46,071	46,042
W.P Labuan	1,388	2,476	1,022	1,108	388
<b>Production value, RM '000 (<math>\approx</math>USD '000)</b>	<b>1,594,762 (<math>\approx</math> USD 449,330)</b>	<b>2,139,347 (<math>\approx</math> USD 602,769)</b>	<b>2,581,546 (<math>\approx</math>USD 727,360)</b>	<b>2,444,684 (<math>\approx</math> USD 688,799)</b>	<b>4,155,391 (<math>\approx</math>USD 1,170,797)</b>

<sup>a</sup> Data retrieved from DOA (2012)

<sup>b</sup> Data retrieved from DOA (2013)

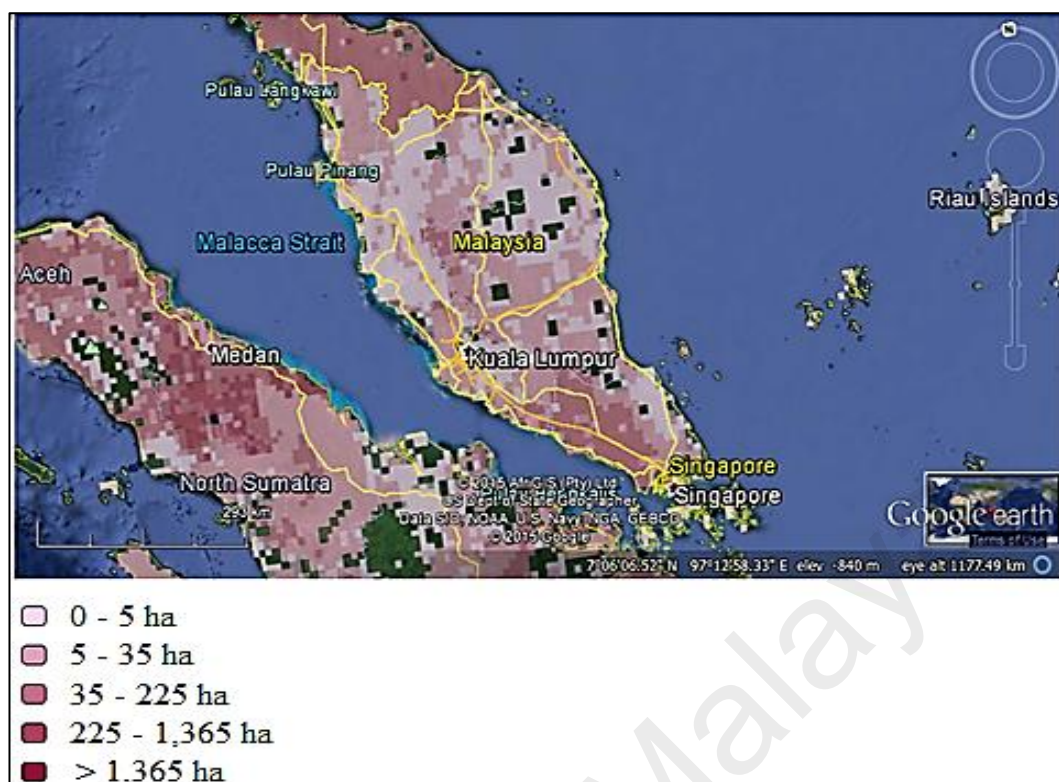


Figure 2.4: A satellite image showing vegetable cropping areas in Peninsular Malaysia.

Credit: IIASA Geo-Wiki Project; Google View

Table 2.4: Vegetables cultivation area and production by state in Malaysia, 2013 (DOA, 2013)

	Planted area, Ha	Harvested area, Ha	Production, tonnes
Johor	11,464	11,187	325,326
Kedah	1,179	1,139	13,789
Kelantan	3,562	3,471	78,283
Melaka	1,624	1,607	16,073
Negeri Sembilan	1,959	1,842	23,042
Pahang	22,799	22,220	615,564
Perak	4,572	4,153	54,269
Perlis	127	68	694
Pulau Pinang	1,043	986	15,438
Selangor	5,412	5,381	81,534
Terengganu	1,204	1,118	14,011
<b>Peninsular Malaysia</b>	<b>54,946</b>	<b>53,171</b>	<b>1,238,024</b>
Sabah	3,468	3,427	42,050
Sarawak	4,490	3,939	46,042
W.P. Labuan	127	124	388
<b>Malaysia</b>	<b>63,030</b>	<b>60,660</b>	<b>1,326,504</b>

## **2.5 Conventional farm system**

Conventional farming is also known as modern agriculture, intensive farming or industrial farming and has contributed tremendously in efficient agriculture production. Global food production increased around 70% to 90% in past decades mainly due to efficient conventional agriculture (Siwar & Hossain, 2001). Currently, there is no single definition of conventional farming. According to USDA, conventional farm system varies according to different farm management systems and the main characteristics are rapid technological innovation, large capital investments, large scale farm, single crops/row crops grown continuously over many seasons, uniform high yield hybrid crops, extensive use of pesticides, fertilizers, water and external energy input, high labour efficiency and dependency of agribusiness (Gold, 2007; Siwar & Hossain, 2001).

Conventional farming method is always seen as polluting the environment due to high-input of fertilizer, pesticide and herbicide. Conventional vegetable farming often involves issues such as over tillage, excessive use of chicken manure and fertilizer, soil salinity, soil erosion, over irrigation, exposed of bare soil to rainfall and thus has the potential to damage soil health, leading to poor productivity and large environmental impacts (Wells *et al.*, 2000). In Malaysia, extensive application of fertilizer and chicken manure is a common practice at conventional agriculture (Tiraieyari *et al.*, 2014).

## **2.6 Organic farm system**

The environmental concern of conventional farming has increased the interest of replacing chemical/synthetic input with organic inputs that is more favourable to the ecosystem function. Agriculture production without synthetic input is the typical concept of organic farming to the general public. Yet, there is a deeper meaning to what it is. Organic farming is a holistic and integrated approach that foster natural ecosystem

processes with the aim to achieve environmental conservation, nutrient cycling, and energy conservation (IAASTD, 2009). The International Federation of Organic Agriculture Movements (IFOAM), defined organic agriculture as (IFOAM, 2015a),

“Organic agriculture is a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic Agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved.”

The core of organic farming is based on four principles: the principle of health, the principle of ecology, the principle of fairness and the principle of care (IFOAM, 2015b). The fundamentals of organic farming is to promote biological activity, nutrient cycling and good soil structure in order for plant to be pest and disease resistant (IAASTD, 2009). Organic farming practice integrated management of traditional, scientific, innovative, and the understanding of ecosystem function to establish a sustainable relationship between environment and human needs. Organic agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved. Some of the common organic practices include:

- Select resilience and disease resistance crops that are suitable for local climate and condition
- Crop rotation
- Intercropping
- Composting
- Certified foliar spray or mineral rock

- Certified bio-pesticides or pest-repellent, and
- Mechanical barrier.

In 2013, there was a total of 43.1 million hectares of organic agriculture land worldwide (Willer & Lernoud, 2015). According to IFOAM statistics, the global organic market was approaching 72 billion US dollars sales in 2013 (Arbenz *et al.*, 2015). Australia/Oceania has the largest organic cultivation area which was 17.3 million hectares (40% of total world organic land), this is followed by Europe with 11.5 million hectares (27% of total world organic land), Latin America (15%), North America (7%), Africa (3%) and Asia (0.7%) (Table 2.5).

Table 2.5: Global total organic farming acreage

Region	Total organic agriculture land in 2013, Ha	Percentage of total organic agriculture land	
		2003	2013
Africa	1,227,008	1.00%	2.80%
Asia	3,425,939	2.60%	8.00%
Europe	11,460,773	22.60%	26.60%
Latin America	6,611,636	20.80%	15.30%
North America	3,047,710	6.70%	7.10%
Oceania	17,321,733	46.30%	40.20%
Total	4,309,113	100.00%	100.00%

Source: Yussefi & Willer, 2003; Willer & Lernoud (2015)

In the past few decades, organic farming in Asia has transformed from fringe farming into a popular farming method due to a higher financial return. In 10 years' time, the share of organic agriculture land in Asia has increased from 2.6% to 8.0% (Table 2.5). However, the area under organic management in Asia region is still considered low as compared to Oceania and Europe (Yussefi & Willer, 2003). The organic market at Asia

is expected to grow due to consumer awareness and concerns on food safety and lucrative financial return. Coupled with the establishment of the ASEAN Standard for Organic Agriculture (ASOA) in 2014, this will boost the regional import-export organic market in ASEAN (FAO *et al.*, 2012).

Organic farming in Malaysia was established in 1986 by the Center for Environment, Technology and Development (CETDEM) for cancer patients who required strict diet (Tiraieyari *et al.*, 2014). Most of the organic farms in Malaysia rely on the use of Bokashi, compost and crop residues to fertilize soil and their major produce was vegetables and only few are growing fruits (Farahzety & Aishah, 2013). In 2009, organic fruits and vegetables were one of the high value agriculture activities that contribute to 1% of GDP Malaysia. In the Tenth Malaysia Plan 2011-2015, special focus on high value activities was given in order to achieve 2% GDP by year 2015 (EPU, 2010). The organic industry in Malaysia was worth RM800 million ( $\approx$  USD 187 million) in 2010 (Meng Yew, 2011). The socio-economic development in Malaysia has led to an increase of consumer's health consciousness and thus increased the demand for organic food (Rezai *et al.*, 2011). Research showed that Malaysians like many others, perceived organic food as more healthy and environmental friendly. However, the price of organic food that is four to five times higher compared to conventional food, is the main barrier for Malaysian consumer towards organic food consumption (Ahmad, 2001). Thus, majority of the organic produce are exported to Singapore which has higher purchasing power. Almost 70% of organic products in Singapore are imported from Malaysia and has encouraged Malaysian farmers to be involved in organic farming especially in southern region like Johor and Negeri Sembilan (Ahmad, 2010)



In 2001, there was a total of 27 organic producers with a total of 131 hectares organic agriculture land in Malaysia (Table 2.6) (FAO, 2004; Razak & Roff, 2007). Currently, the organic producers number has increased to 119 with 603 hectares of organic agriculture land (Willer & Lernoud, 2015). The organic agriculture land in Malaysia is expected to grow continuously. At the same time, there are several challenges that hinder the expansion of organic industry in Malaysia. Below are some of the challenges faced by organic agriculture practitioner:

- Lack of knowledge transfer and training
- Competitive pricing and marketing
- Complex and slow certification process
- Gaining international recognition for organic certification-SOM
- Lack of governmental support, and
- Higher labour demand (Ahmad, 2001).

Table 2.6: Number and area of organic producers per state in Malaysia

State	Number	Area (ha)
Selangor	4	10.8
Negri Sembilan	10	90
Melaka	2	1.1
Johor	2	3.5
Pahang	6	11.6
Sabah	2	12
Sarawak	1	2
<b>Total</b>	<b>27</b>	<b>131</b>

Source: Razak & Roff, 2007

## 2.7 Farm certification

In Malaysia there are two types of certifications commonly applied by the vegetable farmers, namely Malaysia Good Agriculture Practice (MyGAP) and Malaysian Organic Certification Program (SOM).

### **2.7.1 Malaysia Good Agriculture Practice (MyGAP)**

Malaysia Department of Agriculture (DOA) recognises the need to develop sustainable agricultural crop production systems that can secure sufficient food productivity, lessen the impact to environment, and ensure minimum income for the farmers. Thus in year 2002, DOA initiated certification of “Skim Amalan Ladang Baik” (SALM) based on Malaysian Standard MS1784:2005-Crop Commodities. The certification is later rebranded into Malaysia Good Agriculture Practice (MyGAP) on 28 August 2013, which combines the Good Agricultural Practices (SALM), Livestock Farm Practices Scheme (SALT), and Malaysian Aquaculture Farm Certification Scheme (SPLAM) under a single certification process (PEMANDU & EPU, 2013). The MyGAP is benchmarked against ASEAN GAP and Global GAP standard and the certification is aimed to (MPC, 2014):

- increase local consumer confidence in local produce quality
- enhance the competitiveness of local produce
- facilitate export of local produce

Under the new MyGAP certification, a total of 780 farms were certified in year 2014 resulting in a total of 3,200 certified farms since the launch of SALM (PEMANDU & EPU, 2014).

### **2.7.2 Malaysian Organic Certification Program (SOM)**

The increased number of organic farms in the country has captured the attention of Malaysian DOA which initiated the establishment of rules and regulations to prevent farmers from misusing the word “organic”. In 2003, Malaysian Organic Certification Program (SOM) was launched by DOA based Malaysian standard MS1529:2001, the Production, Processing, Labelling and Marketing of Plant Based Organically Produced

Foods (DOA, 2007; Kala *et al.*, 2011). The Malaysian Organic Standard is based on FAO/WHO Codex Draft Guidelines for the production, processing, labelling and marketing of organically produced food (Farahzety & Aishah, 2013). The DOA operates as a regulatory body that monitors local organic produce to ensure the produce are in accordance to standard and in the same time promotes the concept of organic farm (Razak & Roff, 2007). According to the Malaysia Productivity Report 2013/2014, there were 49 out of 89 farmers who were certified organic in 2013 (MPC, 2014).

The SOM certification is not recognised internationally which restricts the local produce to be exported. This leads to some larger commercial organic farms opted for international organic certification such as National Association for Sustainable Agriculture (NASAA) from Australia, The National Organic Program (NOP) from United States, and Japanese Agricultural Standard (JAS) for organic produce. Currently, only Zenxin Organic Food, Loh's Organic Veg-Garden and Titi Eco Farms are certified under NASAA (Meng Yew, 2011).

## **2.8 Environment impact of agriculture**

The increased human population has lead to an increase in the demand of food production. To satisfy the demand, it is estimated that the global food production will increase by 60% (FAO, 2014b). The expansion of agriculture production is estimated to cause 13 million hectares of forests facing land conversion while one third of global farm land is degraded and almost 75% of crop genetic diversity has been lost (FAO, 2015c). With the current situation on land scarcity, soil degradation, reduced biodiversity, and nutrient loss; the extra food production will aggravate the condition (Stoate *et al.*, 2001). Intensive agricultural activities often associate with significant environmental impacts such as deforestation, pollution, greenhouse gas emission, affect

soil quality and reduced biological activity which has compromised food production, environment and social safety. In addition, the competition for water and land are also the major concerns of intensified agriculture production (Willenbockel, 2014). Improper farm practices are the main reason of environmental pollution. Practices such as over tillage, excessive synthetic input (herbicide and pesticide), excessive fertilization, and intensive irrigation can result in severe soil structure damage, soil erosion, soil nutrient depletion, loss of organic matter, nutrient loss and chemicals pollutants contaminate soil and water bodies (Table 2.7) (Killebrew & Wolff, 2010).

Table 2.7: Environment impact of farm practices

Farm practices	Environment impact on			
	Soil	Water	Air	Biodiversity
Mono-cropping				- Reduce biodiversity
No fallow or short fallow period	- Soil nutrient depletion			- Reduce biodiversity
Excessive tillage	- Erosion - Loss of organic matter via decomposition		- Increase soil organic matter decomposition - Contributes to CO <sub>2</sub> emission	
Inorganic fertilizers	- Soil acidification - Nitrate leaching - N loss	- Excess nutrient enter waterways - Eutrophication	- Smog emissions - Ozone emissions - Acid rain - N <sub>2</sub> O emissions	
Pesticides and herbicide	- Accumulate in soil	- Excess chemical contaminate water bodies		- Harms animal and insect - Food safety issues
Irrigation systems	- Salinization (under-irrigate or high saline content water) - Waterlogging (over-irrigate) - Waterlogging (over-irrigate)	- Contamination of water bodies - over-drafting of water		

Source: Killebrew & Wolff (2010)

In long-term these effects can reduce productivity and profitability of farmers due to degradation of the soil and off-site environmental damage (Wells *et al.*, 2000). There are several challenges faced in the efforts to reduce environmental impacts of agricultural sector (Siwar & Hossain, 2001):

- i. Complex and dynamic interactions between farm system, ecosystem, atmosphere and economic,
- ii. Association between environmental issues, social, economic and political welfare, and
- iii. Local environment problems are often overshadowed by global environment issues that attract attention from various NGOs, activists, environmentalists, scientific community and policy maker.

Developed country, such as UK, understands the need to establish legislation and enforcement to regulate farming activities that cause pollution. Thus, more stringent environmental regulations specific for agriculture such as Water Resources (Control of Pollution) (Silage, Slurry and Agricultural Fuel Oil) (England) Regulations 2010 and Nitrate Pollution Prevention Regulations 2015 (SI 668 - 2015) that regulates nitrate release from agriculture land located at sensitive area (DEFRA, 2009, 2013) were launched. Even though agriculture is an important sector in Malaysia, specific legislation and regulations dedicated to agriculture are limited. There are several regulations under the Environment Quality Act 1974 that is pertaining to agricultural activities, but mainly on effluent discharge and open burning activities (DOE, 1977, 1978, 1979, 2009). The regulations are:

- i. Environmental Quality (Prescribed Premises) (Crude Palm-Oil Regulations, 1977),
- ii. Environmental Quality (Prescribed Premises) (Raw Natural Rubber),

- iii. Environmental Quality (Industrial Effluent) Regulations 2009,
- iv. Environmental Quality (Sewage and Industrial Effluents) Regulations, and
- v. Environmental Quality (Clean Air) Regulations 1978.

### **2.8.1 Environmental challenges of organic farms, conventional farms and vegetable farms**

Conventional farm is usually labelled as unsustainable as it is often associated with improper farm practices such as over tillage, excessive use of fertilizer and pesticide and exploitation of soil (Killebrew & Wolff, 2010). Organic farming is often thought of as being more environmental friendly compared to conventional farm due to its characteristic low nutrient intensity, high organic matter nutrient regime, and no agrochemical use (Bavec & Bavec, 2014; Campanelli & Canali, 2012). The high use of organic input in organic system was able to improve soil health and quality, return of C to soil, reduced nutrient losses, and enhance agri-environment biodiversity (Lynch *et al.*, 2012). However, there is also potential nutrient loss from the organic system due to excessive nutrient enrichment (Hartz, 2006). Repeated application of manure and compost in organic system will lead to build up of nutrient content that can easily be lost to environment (Wyland *et al.*, 1996). Assimilation of nutrient from compost or manure may not be synchronized with the crop planting cycle which results in excess nutrient at the root zone that vulnerable to leaching (Evanylo *et al.*, 2008). Another main concern in organic farming is the risk of depleting soil nutrient reserves due to a lack of approved nutrient sources available (Goulding *et al.*, 2008).

A typical vegetable production requires high fertilizer application and frequent irrigation because vegetables, especially leafy types, has shallow roots which limit the water and nutrient use efficiency (Hartz, 2006; Huang *et al.*, 2006a). Thus, vegetable

farming are often associated with low efficiencies of nutrient use and large nutrient losses (Agneessens *et al.*, 2014). Vegetable farming usually practiced extensive tillage with no cover crops, thus prone to erosion. Vegetable farming has a major environmental challenge especially in nutrient management.

### **2.8.2 Agriculture Waste/Residue**

In general, agricultural waste is defined as waste produced by agriculture activity and agriculture premises. The typical examples are livestock waste, crop residues, packaging materials (fertilizer, compost, pesticide and herbicide packaging), and hazardous waste (oil, lead acid batteries and agrochemical) (NIEA, 2012). In Malaysia, there is no definition on the term “agricultural waste” in any Act of Malaysia (Ishak & Samah, 2010). According to OECD, agricultural waste generated from agricultural operations which includes manure, harvest waste, runoff, pesticide and/or fertilizer that enters water, air, soil, salt and silt drained from the farms (OECD, 2001).

Before the enactment of the Environment Quality Act 1974, open burning was a typical farm practice to dispose agriculture waste (DOE, 1974). Until today, the legislation, regulations and guidance available for “agriculture waste” are in general term “waste” and not customized for agriculture waste (Mustafa & Ho, 2006). Some of the waste generated from agriculture activities like chemical contaminated plastic, lead acid batteries, scraps and diesel oil contaminated materials are often hazardous which required special handling and disposal. However, due to low quantities of hazardous waste generated from agriculture premises, the agriculture waste is often neglected by policy makers and law enforcer. In addition, the distribution of agriculture areas in Malaysia are often scattered around suburb and remote areas thus increased the difficulties for waste collection and transport. With increased concern for environment

impact and increased trend of biomass 3R, a specific statute on agricultural waste management is much in need in order to avoid any mismanagement of agriculture wastes that leads to any harm to the environment and human health (Koopmans & Koppejan, 1997).

### **2.8.2.1 Agriculture Biomass**

Agriculture biomass waste is the organic fraction generated by agricultural activities. The usual disposal of agriculture biomass such as crop residues and livestock manure is either incinerated or dumped as organic fertiliser through natural decomposition (Ng *et al.*, 2012). Agriculture biomass wastes such as livestock manure and crop residues is valuable soil amendment that improves soil properties, increased organic content of soil with low organic matter, increased water retention ability of soil, supply nutrient (N, P, K and trace elements) to crop, act as buffer against pH fluctuation and increased soil fertility (Eli *et al.*, 1999). Thus, farmers often leave agriculture biomass wastes on field to replenish soil nutrient. However, waste agricultural biomass emits methane and leachate when the biomass starts to rot and decompose which contributed to GHG emission (UNEP, 2009). The recently launched Eleventh Malaysia Plan (RMK-11), Biomass Strategy 2020 National and Waste to Wealth Program has emphasized on reducing environmental impact through 3R of agriculture biomass and at the same time generate extra income for farmers. Biomass resources are converted into high value products like biofuel, composite, plywood, furniture, animal and cellulose which contributes to GHG emission reduction (UNEP, 2012). Yet, there are concerns on removal of organic residues from agriculture soil may lead to nutrient loss and soil degradation that damage the soil structure of the soil and worms disappears.



### 2.8.2.2 Agriculture Plastic Waste

In the past few decades, there is an increased use of plastic materials in agriculture sector which is also known as plasticulture. The plastic materials used for agriculture management purpose are generally known as agriculture plastic (Hurley, 2008; Schrader, 2000). The development of plasticulture results in higher quality produced, increased yields, reduced weeding, control pest and diseases, less use of pesticide and herbicide, reduced soil erosion, increased nutrient use efficiency, clean production, increase production cycle, and conservation of water (Ingman *et al.*, 2015; Kyrikou & Briassoulis, 2007; Lamont, 2005; Schrader, 2000). Generally, agriculture plastic wastes are from agricultural films, irrigation pipes and fittings, agrochemicals packaging, fertilizer bags and other agricultural plastic (Table 2.8) (Lamont, 1996).

In Europe, the use of plastic for agricultural activity was 47.5 tonnes of plastics in 2007 (Dimitrijevic *et al.*, 2013). The annual agriculture plastic contributed 2% of total plastic consumption in Europe which generated 700,000 tonnes of waste every year (Briassoulis *et al.*, 2013a). In 2008, 1.2 million tonnes of agricultural plastic waste was generated in EU-27, Norway and Switzerland. Of this total amount, 0.67 million tonnes was disposed of (53.6%) and 0.58 million tonnes was recovered (46.4%) (Mudgal *et al.*, 2011). In 2002, it was estimated 0.76 billion kg of plastic was used in the US agricultural sector. A survey in California, US on plasticulture indicates 23% to 93% of farmers used agriculture plastic and only 13% to 46% was recycled (Hurley, 2008). The most common treatment for agriculture plastic are burned, buried in soil, and discarded in fields or landfilled (Sonnevera, 2011). Burning of agriculture plastic or burial in soil causes irreversible environment damage such as soil contamination, soil degradation, GHG and sot particles emission, and dioxin and furan emission. All these have negative impacts to human health.

Table 2.8: General plastic product and usage, estimated waste generation range and waste composition in Spain, Greece and Italy

Plastic Product	Purpose	Estimated quantity of waste (kg/ha/yr)*	Composition of plastic
Agricultural films	Greenhouse	463-1515	LDPE, LLDPE, EVA, copolymers, PVC, PP, HDPE
	Low tunnel	280.5-975	
	Mulching	106 - 335	
	Direct cover	NA	
	Silage	1015	
	Bale wrap	NA	
	Shrink wrap	NA	
Pipes and fittings	Irrigation system	16-140	LDPE, HDPE, ester, PE, PVC, PP
	Hydroponic system	NA	
Container	Agrochemical, herbicide, pesticide	0.5-3.0	PET, LDPE-HDPE, PA, PBT, PP, PVOH, EVOH
Packaging bag	Fertilizer compost	1.5-4.5	PE, PP
Net	Cover	500	HDPE, PP, LLDPE

Source: Mudgal et al., 2011; Briassoulis et al., 2012; Briassoulis et al., 2013a

EVA: ethylene vinyl acetate,

EVOH: ethylene vinyl alcohol,

HDPE: high density polyethylene,

LDPE: low density polyethylene,

LLDPE: linear low density polyethylene,

PA: polyamide ,

PE: polyethylene,

PP: propylene,

PBT: polybutylene terephthalate,

PVC: polyvinylchloride ,

PVOH: polyethanol,

NA: Not available

In Italy, Greece and Spain, the recovery and recycling rate of pesticides, fertilizers or seed packaging are up to 60% and 49.5% (Table 2.9) (Briassoulis *et al.*, 2013b). The collections of agriculture plastic are difficult especially in remote areas and it is more challenging when the transport cost is high due to the long distance and bulky nature of the waste (Briassoulis *et al.*, 2013b; Hurley, 2008). In addition, limited recycling facilities and restrictions on disposal at municipal landfill are some of the reason for improper waste management in farms (Sonnevera, 2011). Other than that, some farmers prefer to burn or bury the agriculture wastes accumulated in farms due to lack of

education and information on waste management. The exact data of illegal disposal of plastics wastes by farmer is not available.

Table 2.9: Agriculture plastic waste recycling in Italy, Greece and Spain

Country	Year	Recycling Volume
Italy	2005	- 95 kt of agricultural films - 45 kt of irrigation pipes, agrochemicals packaging and other agricultural plastic
Greece		- 0.8 to 1.0 kt/yr agricultural plastic waste
Spain	2009	- 482.9 kt/yr of plastics (in general)
EU28+2	2014	- 5% of total plastic waste from agriculture - 28.0% (26.4%) of this was recycled, while

Source: Briassoulis et al., 2013b and EPRO, 2015

### 2.8.3 Deforestation

Commercial agriculture is the key driver of deforestation and it is estimated that 50% of the global deforestation in the last few decades have been for agriculture purpose (Lawson *et al.*, 2014). According to FAO, several countries have achieved or almost reached the limits of available land especially in South Asia and East & North Africa (Alexandratos & Bruinsma, 2012; FAO, 2009). Moreover, there is an increased interest in biomass energy and agricultural land used for urban constructions which will worsen the situation of competition for land (Willenbockel, 2014). Deforestation poses one of the gravest threats to biodiversity where up to 75% of the genetic diversity of crops has already disappeared (Bruinsma, 2003). Deforestation also generates nearly 50% more greenhouse gases than the global transportation sector (6.7 Gt CO<sub>2</sub>) (IPCC, 2014b; Lawson *et al.*, 2014).

Oil palm in Southeast Asia is often cited as the major contributor to deforestation and habitat disturbance and Malaysia is one of the world leading palm oil producer (Alexandratos & Bruinsma, 2012). The UNESCAP statistics show that Malaysia's

forest land area has decreased from 68.1 % in year 1999 to 61.7 % in year 2012 (UNESCAP, 2014). The decreased available of land for agriculture has highlighted the importance of increased farm efficiencies by ensuring sufficient food supply are produced with the limited amount of land.

#### **2.8.4 Climate change**

Agriculture has significant effects on climate change, primarily through land use change and greenhouse gases emissions (Willenbockel, 2014). It is one of the major driver and source of greenhouse gas emission (FAO, 2014b). Research suggest from 1961 to 2010, the CO<sub>2</sub> emission from intensive agriculture increased about 15% (Zeng *et al.*, 2014). The relationship between climate change and agriculture are interrelated. Climate change affects global temperature, precipitation, change of climate pattern, increase CO<sub>2</sub> concentration in the atmosphere (Beddington *et al.*, 2012b). It is projected that in next few decades there will be an increase of 4.0 to 5.8 °C of global temperature (Chauhan *et al.*, 2014). All these would most likely affect the current agricultural productivity as crop yield is highly sensitive to climate. However, research shows that the impact of climate change on agriculture production may not be as serious as predicted (Chauhan *et al.*, 2014; Long *et al.*, 2006). This is because the crop yield loss caused by rise in temperature and lower soil moisture is counter balanced by the increase yield due to direct fertilization effect of increased CO<sub>2</sub> in the atmosphere. Thus, the overall effect of climate change on agriculture will depend on the balance of these effects. But the increased occurrence of extreme weather events such as droughts, floods, frosts, soil salinisation due to increased sea level and heat waves will no doubt cause damage to agricultural output (Beddington *et al.*, 2012a). An economic perspective model shows that the climate change caused four times greater economic loss compared to other environmental stresses (Adams *et al.*, 1990). There is an urgent need for policy

makers, industry player, farmers, and civil society to formulate plans for adapting agriculture to climate change (Howden *et al.*, 2007; Smit & Skinner, 2002; Vermeulen *et al.*, 2012). In order to adapt agriculture to climate change, the understanding of farm system metabolism is crucial as farming system varies locally due to different climate, farm practices, soil and ecosystem.

### 2.8.5 Greenhouse gas emission (GHG)

About 13.5% of global GHG emissions are due to crop and livestock production (FAO, 2014b). Almost half of the global greenhouse gases is contributed by N<sub>2</sub>O emission from agricultural systems (Yang *et al.*, 2014a). In the effort to achieve food security and sustainable food production, greenhouse gas emission caused by land use is a major challenge (Willenbockel, 2014). From 1992 to 2012, the global GHG emission from agriculture has increased about 18.4% and China is the number one GHG emitter (Figure 2.5).

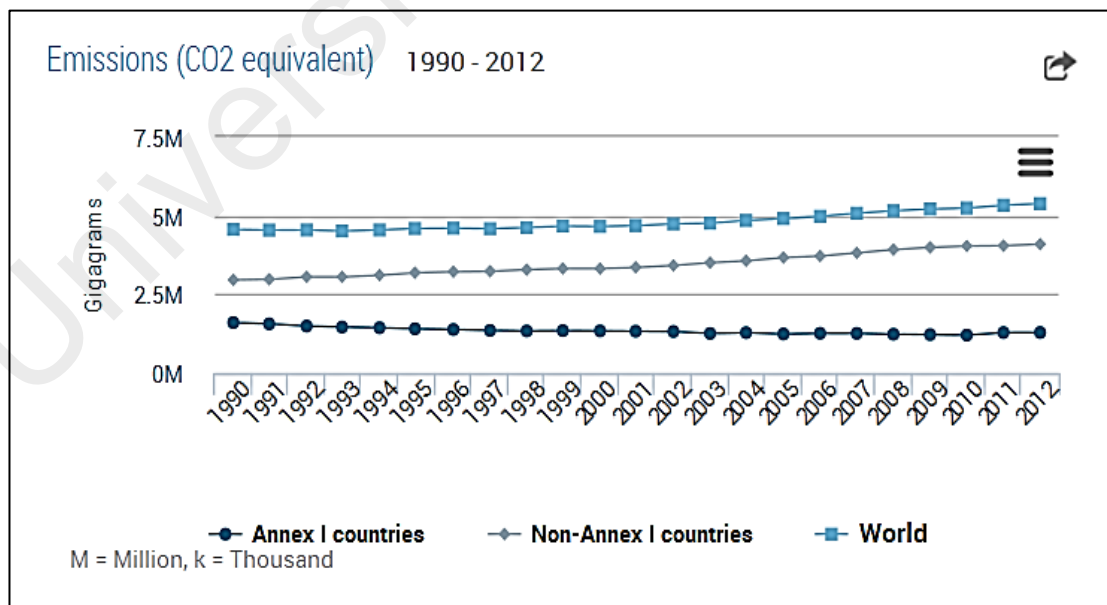


Figure 2.5: Worldwide greenhouse gas emission from agriculture (CO<sub>2</sub> equivalent),

1990-2012 Source: (FAO, 2015b)

Agriculture sector is the largest contributor to non CO<sub>2</sub> emissions, for instance, in year 2005 about 82% of non CO<sub>2</sub> emissions in Central and South America was contributed by agriculture sector (Figure 2.6). The trend for global GHGs emissions by agricultural sector between 1990 and 2010 implies there will be continuous increase of GHG emission in the future (Figure 2.7). The rapid increase of fertilizer and pesticide usage, increase in agriculture production and thawing peat land in the pass decades have escalated the GHG emission in recent years (Wu & Ma, 2015).

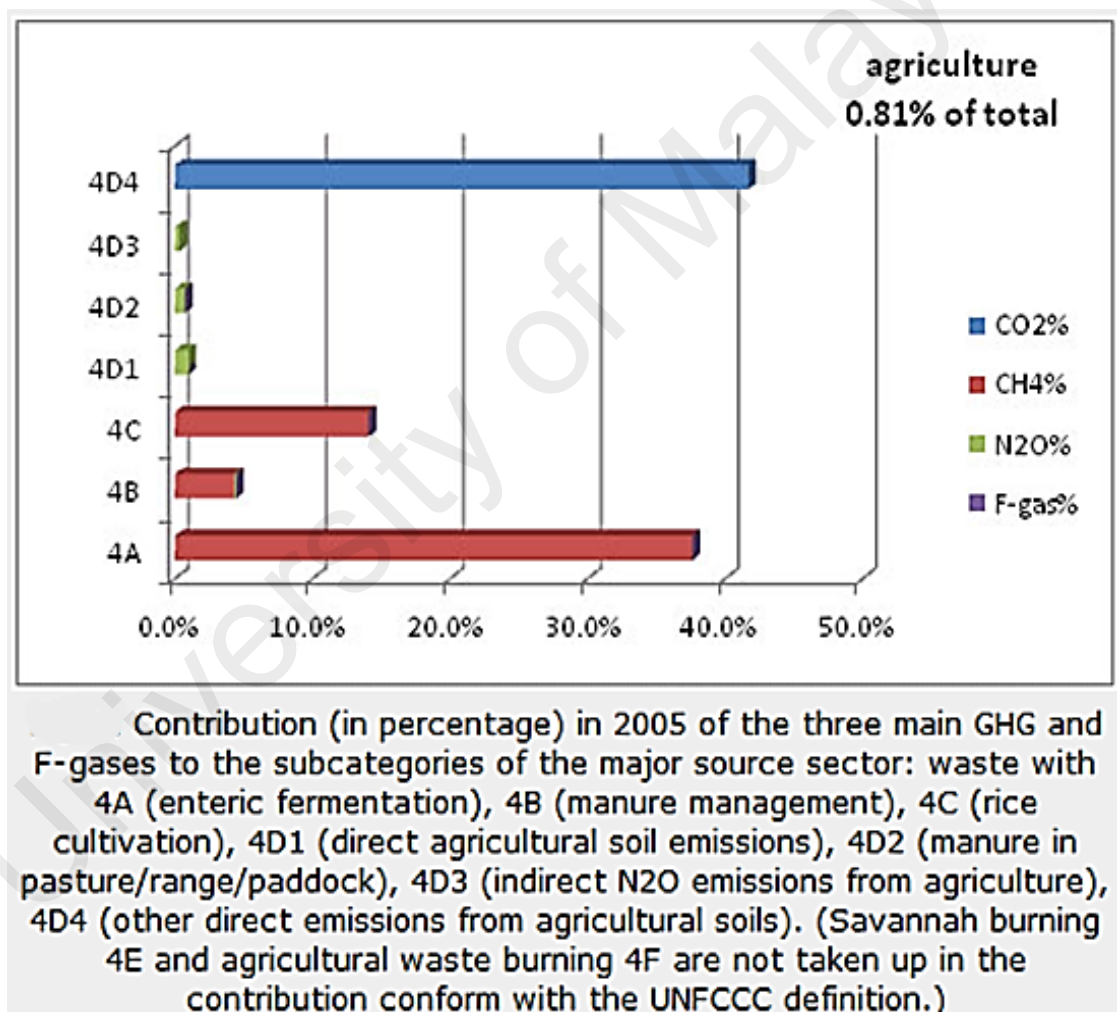


Figure 2.6: The three major gases emitted from agriculture land worldwide (Janssens-Maenhout *et al.*, 2011)

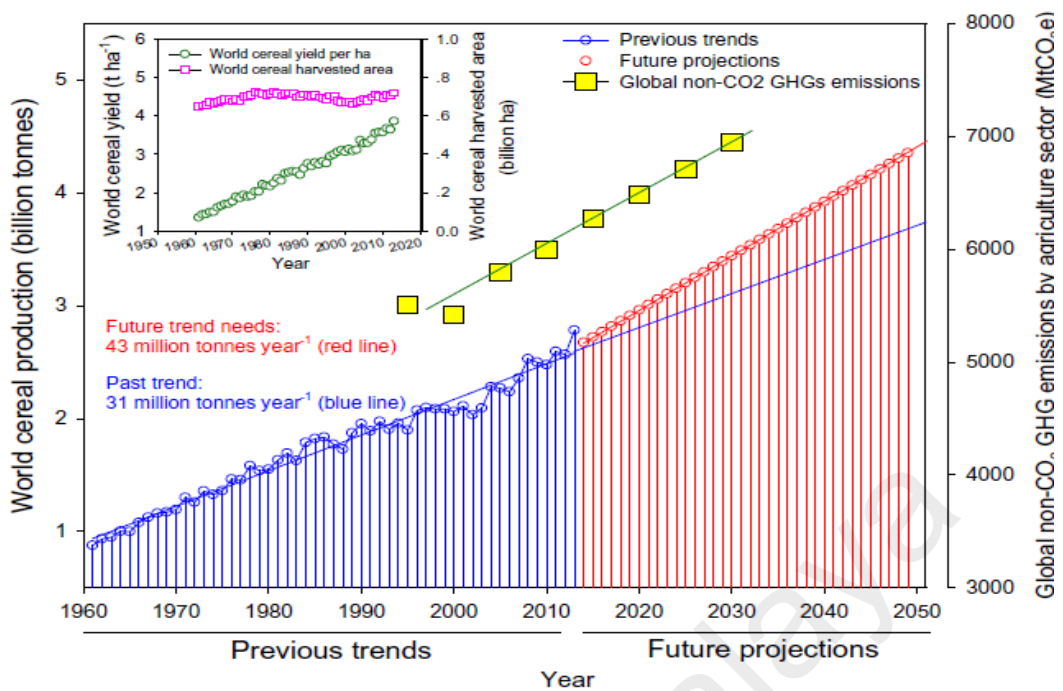


Figure 2.7: World cereal production targets and global non-CO<sub>2</sub> greenhouse gas (GHG) emissions by agriculture sector (Wu & Ma, 2015)

There are several pathways for gaseous emission from agriculture soil (Table 2.10). Soil respiration is the primary pathway of CO<sub>2</sub> loss from agriculture soil due to root and microbial activity (Schlesinger & Andrews, 2000). Soil disturbance, soil tillage, grazing activities and land use change are also the main factors of increases soil respiration (Tilman, 1999). Other than increase of temperature, soil organic matter and soil moisture will also favour the biological fixation. Thus, the agriculture soil in tropical region is found to have higher rate of soil respirations than temperate region due to higher temperature and humidity (Schlesinger & Andrews, 2000). In addition to these, the use of N fertilizer, tillage, manure, crop residues and biomass burning also contributes to a higher emission of nitrous oxide, ammonia, carbon dioxide, and methane (Figure 2.8) (Bashir *et al.*, 2013).

Table 2.10: Gaseous emission pathway from agriculture soil

Gaseous	Sources from agriculture activities
Nitrous oxide and ammonia volatilisation	Nitrification and denitrification processes in agricultural soils and manure. The N leached from agriculture land to water courses may contribute to gaseous emission.
Methane	Enteric fermentation of livestock and anaerobic decomposition of organic matter.
Carbon dioxide	Decomposition of organic matter, liming of agricultural soils and combustion.

Source: Yli-Viikari et al. (2007)

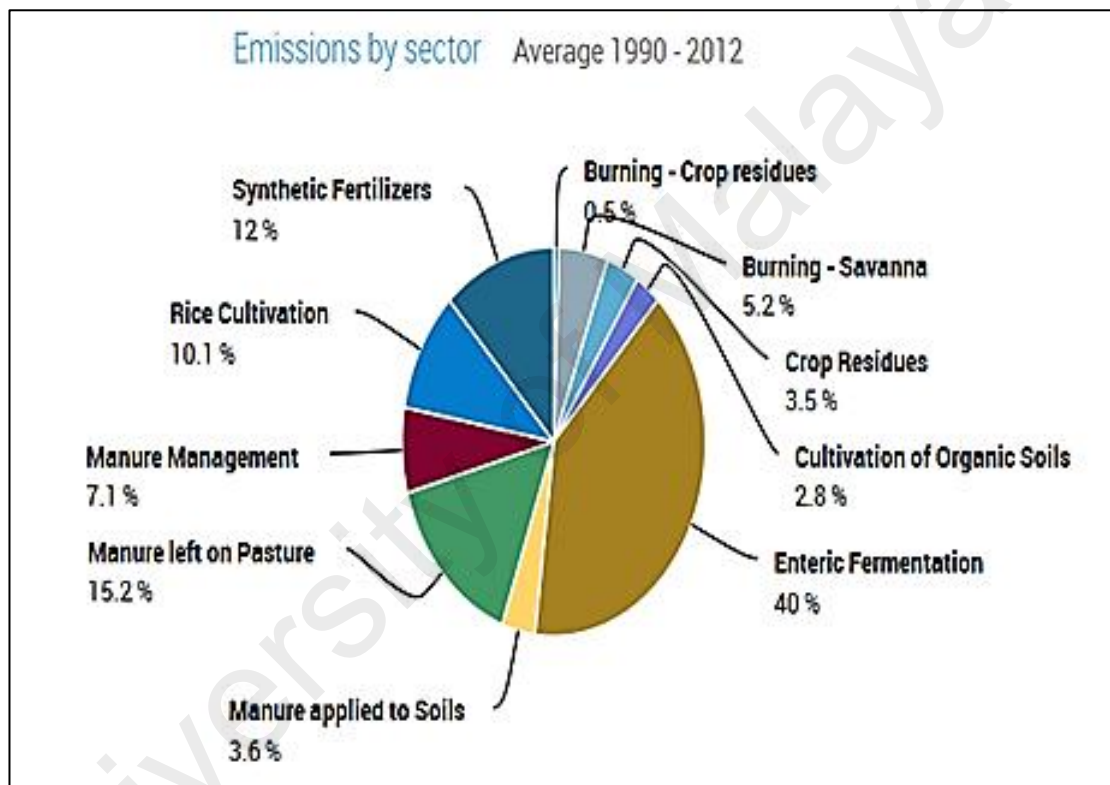


Figure 2.8: Global share of greenhouse gas emissions from agriculture by sector, 1990-2012 (CO<sub>2</sub> equivalent) Source: (FAO, 2015b)

Crop residues are major agriculture waste and it is usually left on field as soil amendments. The organic matter inputs via crop residues may lead to GHG emission. Thus the effort to enhance carbon sequestration by increasing biomass input into agriculture soil appears to cause the emissions of N<sub>2</sub>O (Nadeem *et al.*, 2015). It is estimated that the oil palm residues in Malaysia has the potential to release 4.19 million



tonnes of CO<sub>2</sub>E between year 2011 and 2032 (EPU, 2015; UNEP, 2012). Livestock manure is another major farm input as a cheaper alternative of nutrient source. However, the decomposition of livestock manure generates CH<sub>4</sub> especially under anaerobic condition. Improper storage, handling and application of manure are crucial to reduce possibility of anaerobic decomposition which then reduced CH<sub>4</sub> emission rate (DEFRA, 2014a).

The emission of N<sub>2</sub>O, CO<sub>2</sub>, and CH<sub>4</sub> not just contributes to global warming but also detrimental to environment, animal and human health. The NH<sub>3</sub> emissions not only affect air quality and if the emission area is located near to sensitive zone, this will also cause eutrophication and soil acidification. (DEFRA, 2015a). It has been estimated that with proper farm management and mitigation method, the potential to achieve a reduction of 327 Kt CO<sub>2</sub>E of GHG emission is possible (DEFRA, 2014c). Following good farm practices are suggested to lower the operational cost of farms by preventing unnecessary loss of nutrient (DEFRA, 2014a, 2014c):

- improve the efficiencies and effectiveness of nutrient use
  - proper manure management
  - protect and enhance soil carbon stock
  - appropriate timing of fertilizer application
  - minimal tillage, and
  - maintaining soil pH within the neutral range able to reduce N<sub>2</sub>O emissions
- (Nadeem *et al.*, 2015).

### **2.8.6 Water resource**

The volume of renewable water available each year per person in Asia and the Pacific has reduced between 1992 and 2012 (Table 2.11) (UNESCAP, 2014).

Table 2.11: Water source and water withdrawal in Asia Pacific

	1992	2012	% change
Renewable water available, m <sup>3</sup> per capita per annum	30,200	19,836	-34%
Domestic water withdrawal, m <sup>3</sup> per capita per annum	44.9	151.0	+236%

Source: UNESCAP (2014)

Globally, about 70% of water is used for agricultural activities while in Southeast Asia about 92% of water usage is for agriculture sector (Figure 2.9) (FAO, 2015a; UNESCAP, 2014). Similar with land availability, the South Asia and East & North Africa regions are under the pressure on renewable water resources for irrigation (FAO, 2009). There is a tension between countries over water source as over one fifth of water resources are shared globally (Beddington *et al.*, 2012b).

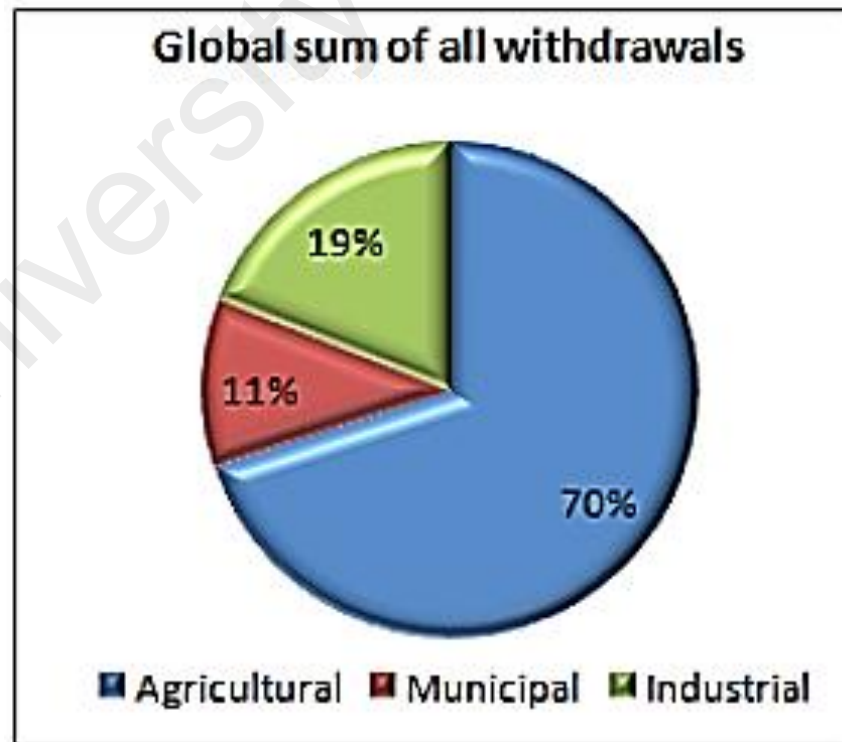


Figure 2.9: Global water withdrawal (AQUASTAT) (FAO, 2015a)

Besides high volume of water usage, agriculture activities are also one of the major sources of pollution to surface and underground water. The main pollutants from agriculture land are nutrients, pesticides, herbicide, sediment, antibiotics and faecal bacteria (Ribaudó *et al.*, 1999; Zainudin, 2010). The excessive use of agrochemical and manure are washed off from the agriculture land by rainfall and irrigation and enter the water bodies as runoff and leachate (Figure 2.10) (DEFRA, 2015a).

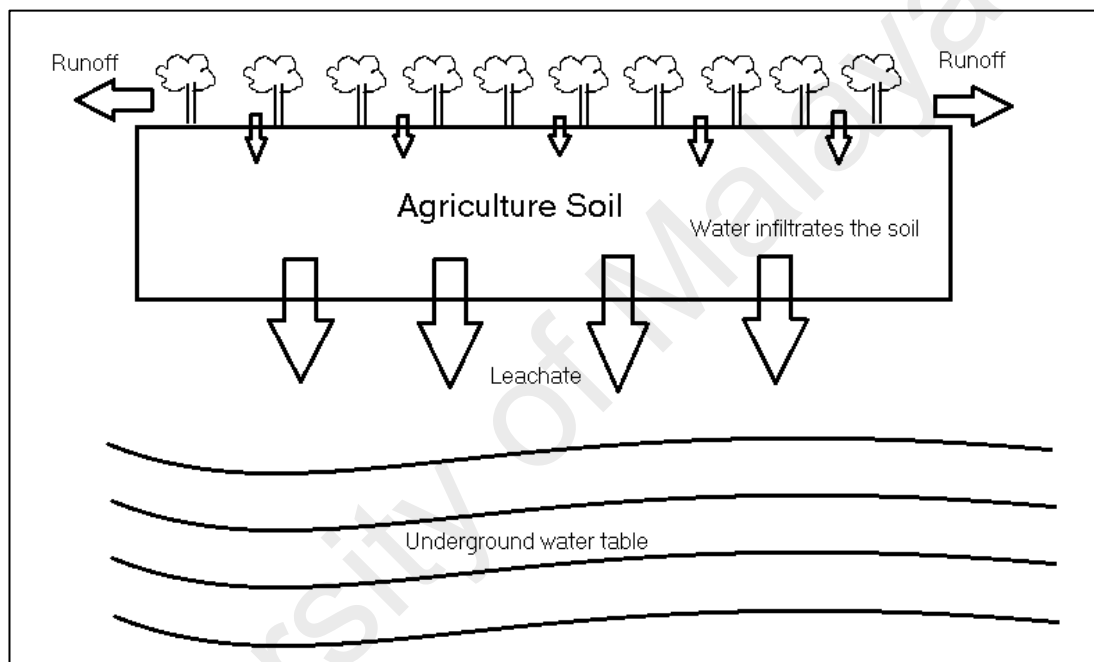


Figure 2.10: The flow of runoff and leachate from agriculture soil

The nutrients in water ways can cause eutrophication, algal bloom and the pathogen attached on wash off manure or soil can easily contaminate the water bodies (Beddington *et al.*, 2012b). Several incidences of cyanobacterial bloom occur in drinking water and recreational water which affects peoples and even has caused death (Table 2.12 and Table 2.13)

Table 2.12: Global cases of toxic cyanobacterial blooms reported in drinking waters

(Bláha *et al.*, 2009; WHO, 2012)

Location and date of Cyanobacterial bloom	Species	Symptoms	Consequences
USA, 1931	<i>Microcystis</i>	Gastro-enteritis	No data
USA, 1976	<i>Schizotrix, Plectonema, Phormidium, Lyngbia</i>	Gastro-enteritis	62% of the population fed by the network become ill
Australia, 1979	<i>Cylindrospermopsis raciborskii</i>	Hepatitis	141 hospitalizations
Australia, 1981	<i>Microcystis</i>	Gastro-enteritis, liver injury	No data
Brazil, 1988	<i>Anabaena, microcystis</i>	Gastro-enteritis	2000 people affected 88 deaths
Sweden, 1994	<i>Planktothric agardhii</i>	Gastro-enteritis	121 people affected
Brazil, 1996	<i>Aphanizomenon, Oscillatoria, Spirula</i>	Hepatitis	166 people affected 66 deaths
China, 1977-1996	<i>Microcystis</i>	colorectal cancer, deaths	No data

Table 2.13: Cases of toxic cyanobacterial blooms reported in recreational waters

(Giannuzzi *et al.*, 2011; WHO, 2012)

Location and date of Cyanobacterial bloom	Species	Symptoms	Consequences
Canada, 1959	<i>Microcystis, Anabaena circinalis</i>	Gastro-enteritis, headaches, nausea, muscular pains	30 people affected
UK, 1898	<i>Microcystis</i>	Gastro-enteritis, vomiting, sore throats	20 people affected 2 hospitalization
Salto Grande Dam, Argentina, 2007	<i>Microcystis spp</i>	Nausea, abdominal pain and fever, dyspnea and respiratory distress	1 people affected and hospitalizations
California reservoirs, 2007	<i>Microcystis aeruginosa</i>	Dermal symptoms, earache, headache, abdominal pain, upper respiratory symptoms,	81 children and adults affected

### 2.8.7 Nutrient loss via runoff and leachate

Non-point source (NPS) pollution of nutrient into water bodies is one of the notorious environment impacts from agriculture land (Kronvang *et al.*, 1995; Kyllmar *et al.*, 2014; Ribaudo *et al.*, 1999; Rozemeijer *et al.*, 2014; Stoate *et al.*, 2001; Wolff, 2003; Woodward *et al.*, 2012). In the past 20 years, there is a rapid increase of synthetic fertilizer and pesticides usage which reveals the potential increase of NPS incident (Figure 2.11). Several studies demonstrated a correlation between the nutrient concentration in surface water with fertilizer application and the runoff from agriculture land (Aweng *et al.*, 2011; Ibrahim *et al.*, 2011; Ismail *et al.*, 2007; Mohd Ekhwan *et al.*, 2012).

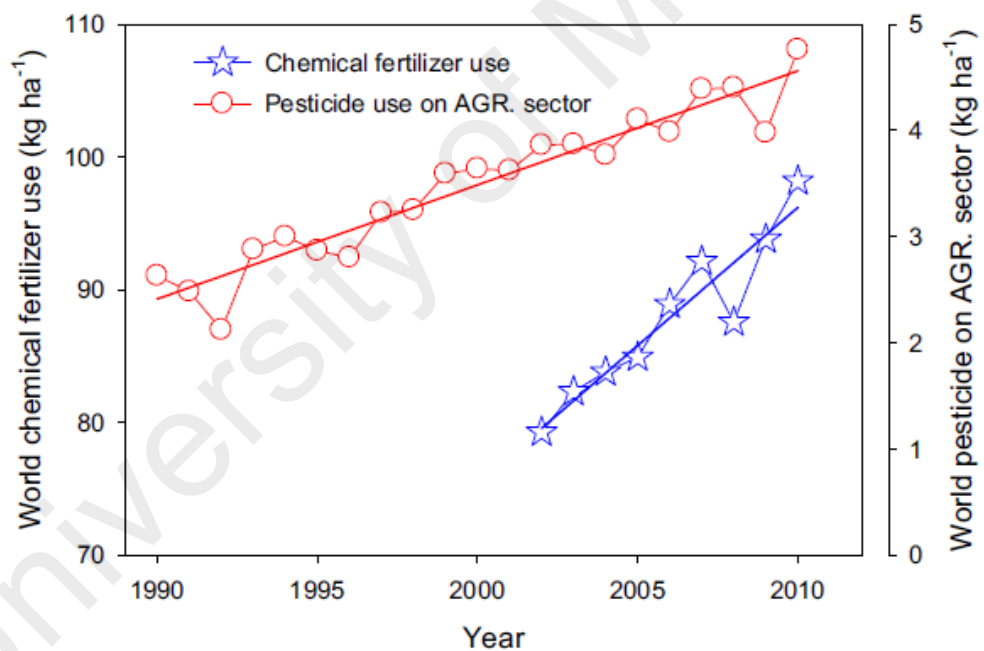


Figure 2.11: World chemical fertilizer and pesticide usage, 1990-2010 (Wu & Ma, 2015)

Crop requires certain amount of nutrient for optimal growth and excess nutrient applied is exposed to runoff and leaching which will transport the nutrient,  $\text{NH}_3$ ,  $\text{NO}_3$ , and  $\text{PO}_4$  into groundwater or surface water. There are two types of nutrients transport from soil

to water: (1) chemicals dissolved in runoff and leachate and (2) chemicals bound on to suspended solid and exported via soil erosion (Cade-Menun *et al.*, 2013).

Runoff and leaching from agricultural lands proximate to surface waters can deteriorate surface water quality (Bashir *et al.*, 2013). It is reported that almost 55% of eutrophication of surface water at EU are due to agriculture and in UK about 59% of nitrates in inland waters originated from agricultural activity (Buckley & Carney, 2013; DEFRA, 2015b). Thus, runoff is identified as the major contributor to nutrient loss and NPS pollution (Nie *et al.*, 2012; WHO, 2012). The runoff and leaching from agriculture soil also contributes to GHG emission (DEFRA, 2015a). Nutrients exited from agriculture land will not necessarily enter the water bodies as it may be retained in the soil particle along the transport route (Bashir *et al.*, 2013; Reidsma *et al.*, 2012). Therefore, transport factors like soil texture, soil permeability, cation exchange capacity (CEC), saturated hydraulic conductivity, slope of cultivation area, distance of agriculture land to water body, riparian buffer zone, irrigation erosion, rainfall, surface runoff, leaching, soil erosion, drainage and biological activity will determine the extent of nutrient loss entering the water bodies (Nie *et al.*, 2012; Pärn *et al.*, 2012). The complex interrelation of farm practice and transport factor has made the effort to mitigate agriculture NPS more difficult and challenging.

Study shows that the nutrient stock in the water catchment is highly correlated with nutrient loss from agriculture land (Xiao-xue *et al.*, 2014). In addition, intensive irrigation and high precipitation increased the runoff and leaching volume which is closely linked to nutrient transport from the soil (Pimentel *et al.*, 1995). It was found that higher nutrient concentration in streams that are near to high intensity farms areas than low intensity farms areas (Bechmann *et al.*, 2008). The farm intensity is highly

dependent on the individual farm management. Therefore, proper nutrient, soil and water management is crucial in reducing the impact on water quality but also the cost of farm input. The nutrient input at organic farm is lower as it depends on organic input, biological fixation, crop rotation, nutrient cycling of crop residues, and nutrient retention via green manure (ADAS *et al.*, 2006; Lynch *et al.*, 2012). Thus, the lower nutrient intensity of organic system is recognised to have less susceptibility to nutrient loss as compared to conventional production.

The contaminated water affects biodiversity and aqua-system through eutrophication, algal bloom, hypoxia which increase the fatality rate of aquatic animal and increased the cost of water treatment (Bouwman *et al.*, 2013). The economic cost of over nutrient application is not just the water treatment cost but also the cost of wasted nutrient input (Buckley & Carney, 2013). Farmer's lack of crop nutrient requirement and soil nutrient knowledge often leads to following inappropriate farm practices as below which causes nutrient loss in farm system:

- over application,
- inadequate application timing of fertiliser,
- over cultivation and disturbance of soils, and
- profit driven fertiliser application to increase yield and income (Bechmann *et al.*, 2008; Buckley & Carney, 2013; Dungait *et al.*, 2012; Reidsma *et al.*, 2012).

Agriculture is one of the main GDP contributor in Malaysia which covers 15% of the total land in Malaysia (DOA, 2012). Research shows that the impact of agriculture to water quality will be more obvious when agriculture is the prevailing activity at an agriculture region (Schröder *et al.*, 2004). Therefore, as a major agriculture country, Malaysian water also encounters the problem of contamination from agriculture runoff

(Ahmad *et al.*, 1994; Eli *et al.*, 1999). Water quality deterioration due to surface runoff from agriculture land was observed at several major lakes in Malaysia (Sharip & Zakaria, 2007). A preliminary study shows almost 60% of 90 lakes in Malaysia are facing eutrophication problem due to runoff contamination from agriculture land (Sharip & Zakaria, 2007; Sharip *et al.*, 2014). The legislation that are enforced to regulate water pollution related to agriculture sector are:

- i. Environmental Quality (Prescribed Premises) (Raw Natural Rubber),
- ii. Environmental Quality (Prescribed Premises) (Crude Palm-Oil Regulations, 1977),
- iii. Environmental Quality (Industrial Effluent) Regulations 2009, and
- iv. Environmental Quality (Sewage and Industrial Effluents) Regulations.

The legislations available in Malaysia limits the release of untreated effluent into water bodies. The regulations are qualitative and not quantitative, thus constant amount of pollution parameters are released continuously into surface water. Parameter standard such as biochemical oxygen (BOD), chemical oxygen demand (COD), total solids, suspended solids, oil and grease, ammoniacal nitrogen, total nitrogen, temperature, and heavy metal level are to be complied before releasing effluent into environment (Table 2.14). Unfortunately, these rules and regulations mainly focus on point source pollution of oil palm, rubber processing mill, and factory. Non-point source pollution such as nitrate pollution from agriculture runoff and leaching are not regulated by any legislation in the country.



Table 2.14: Malaysian effluent standard for industry

	Oil palm mill effluent <sup>a</sup>	Rubber mill effluent <sup>b</sup>			Industrial effluent <sup>c</sup>		Municipal waste water <sup>d</sup>	
	Parameters limit for watercourse discharge of	Parameters Limit for Watercourse Discharge		Parameters Limit for Land Discharge	Standard A <sup>e</sup>	Standard B <sup>f</sup>	Standard A <sup>e</sup>	Standard B <sup>f</sup>
		Latex products	Non-latex products					
Biochemical oxygen (BOD) 3 day, 30°C, mg/l	100	100	100	6,000	-	-	-	-
Biochemical oxygen (BOD) 5 day, 20°C, mg/l	-	-	-	-	20	50	20	50
Chemical oxygen demand (COD), mg/l	400	400	250	12,000	-	-	50	100
Total solids; mg/l	-	-	-	13,000	-	-	-	-
Suspended solids, mg/l	400	150	150	500	50	100	50	100
Oil and grease, mg/l	50	-	-	-	1.0	10	-	10.0
Ammoniacal nitrogen, mg/l	150*	300	40*	900	10	20	-	-
Total nitrogen, mg/l	200*	300	60*	1,100	-	-	-	-
pH	5.0-9.0	6-9	6-9	3.5-8.0	6.0-9.0	5.5-9.0	6.0-9.0	5.5-9.0
Temperature, °C	45	-	-	-	40	40	40	40

\* Value of filtered sample

<sup>a</sup> Environmental Quality (Prescribed Premises) (Crude Palm-Oil Regulations, 1977) Third Schedule [Regulation 16(1)] Subs. P.U. (A) 183/82 (DOE, 1977)

<sup>b</sup> Environmental Quality (Prescribed Premises)(Raw Natural Rubber) Regulations 1978 Second, Third and Fourth Schedule [Regulations 12(1), 12(2) and 14(1)] Subs. P.U.(A) 74/80 (DOE, 1978)

<sup>c</sup> Environmental Quality (Industrial Effluent) Regulations 2009 First Schedule [Paragraph 11(1)(a)] (DOE, 2009 )

<sup>d</sup> Environmental Quality (Sewage and Industrial Effluents) Regulations 1979 Third Schedule [Regulation 8(1), 8(2), 8(3)] (DOE, 1979)

<sup>e</sup> Standard A discharge of effluent into any inland waters within the catchment areas

<sup>f</sup> Standard B discharge of effluent into any inland waters or Malaysian waters

### 2.8.7.1 Nitrate in runoff and leachate

Nutrients can move from soil to water in dissolved form through runoff water or in particulate form that is bound to suspended sediments. These nutrients exist in a variety of chemical forms which include inorganic (e.g.,  $\text{NO}_3$ ;  $\text{PO}_4$ ) and organic (e.g., DNA, protein) forms (Cade-Menun *et al.*, 2013). Nitrate is one of the N compounds that are highly labile and mobile in water which is prone to loss via runoff and leaching. Other N compounds such as  $\text{NH}_4$  is often bound to soil particle while  $\text{NH}_3$  is often found in the top soil layer which is easily volatilized (Pärn *et al.*, 2012; Tilman, 1999). Nitrate pollution of ground water is severe in vegetable production area of US (Hartz, 2006). A study of 94,600 community water systems show 52% of the water systems contain  $\text{NO}_3$  and 1.2% exceeded the concentration limit (Bashir *et al.*, 2013). This raised the concerns on human health as it may pollute the drinking water. It also poses an environmental threat as it may lead to algal bloom and hypoxia. However, the toxicity level of nitrate is still debateable due to lack of evidence of nitrate impact on human health (Schröder *et al.*, 2004). Nonetheless, stringent regulation has been established by USEPA whereby 10 mg/l limit for nitrate concentration is imposed on surface water for municipal use and drinking purpose (WHO, 2011a).

Nitrate loss is highly dependent on the farm management, nutrient use efficiency, soil condition, soil water holding capacity, rainfall, irrigation, and soil nitrogen content (Pärn *et al.*, 2012). The usual form of N loss in runoff is in ammonium form while it is often in nitrate form for leachate N loss (Bashir *et al.*, 2013). Two factors affect nitrate losses from agriculture. First is direct effect that relates to eutrophication which leads to algal bloom toxicity. Second is indirect effect where N loss is linked with atmospheric deposition acidification and denitrification from nitrate to nitrous oxide triggered by dissolved of sulphate and metals (Schröder *et al.*, 2004). There is a high correlation of

nitrogen concentration in surface water with the nutrient loss from agriculture soil (Xiao-xue *et al.*, 2014). Thus, integrated nutrient management by improving nitrogen use efficiency (NUE) will minimize nitrate loss (Lynch *et al.*, 2012).

### **2.8.8 Microbial contaminant in the farm**

*E. coli* outbreak at Northern Germany in May 2011 due to the consumption of organically produced bean sprout lead to more than 4000 illnesses, 800 cases of the hemolytic–uremic syndrome, and 50 deaths in Germany and in 15 other countries (Blaser, 2011; EFSA, 2011; Frank *et al.*, 2011; Rising & Grieshaber, 2011; Sample, 2011). Manure is identified as one of the key sources of microbial contamination in agriculture land (Bonti-Ankomah *et al.*, 2006; Bourn & Prescott, 2002; IFST, 2013; Yiridoe *et al.*, 2005). Organic waste and manure are the primary source of nutrient at organic farm and it is also widely used in conventional agriculture along with synthetic fertilisers (Santamaría & Toranzos, 2003). It is reported that the use of livestock manure to fertilizer crop increased the risk of *E. coli* contamination in organic farm (Bourn & Prescott, 2002; Mukherjee *et al.*, 2007). This poses questions of food safety and whether organic produce are exposed to higher level of microbial contaminant.

Numerous researches compared the microbial population between organic and conventional farm and no significant differences in the prevalence of pathogen population and mycotoxin was found (Franz *et al.*, 2008; Kuhnert *et al.*, 2005; Lairon, 2010; Mukherjee *et al.*, 2004; Winter & Davis, 2006). Other researches showed otherwise where the highest incidence of *E. coli* was found in organic produce than conventional produce (Mukherjee *et al.*, 2007; Oliveira *et al.*, 2010). One research shows that the enteric bacteria in soil amended with organic wastes has propagules densities of  $1.85 \times 10^7$  and  $3.88 \times 10^7$  CFUs/g dry soil which is higher compared to soils

with synthetic fertilizer  $1.08 \times 10^7$  and  $1.94 \times 10^7$  CFUs/g dry soil (Bulluck *et al.*, 2002). This research further concluded that the higher population of beneficial soil microorganisms in soil amended with organic waste helps to reduce pathogen populations. In addition, study by Johannessen *et al.* (2005) shows there was no transmission of *E. coli* O157:H7 on to lettuce planted on contaminated soil (Johannessen *et al.*, 2005). However, a sampling of 179 samples of organic lettuce showed that contamination with pathogenic bacteria does occasionally occur even though the bacteriological quality is classified as good (Loncarevic *et al.*, 2005).

Agriculture soil contains enteric pathogens mostly due to the use of manures and biosolids. The common enteric pathogens found in agriculture soil are *E.coli*, *Salmonella sp.* and *Shigella sp.* The pathogen have the ability to persist in the soil for eight to twelve weeks after fertilizer application (Gorski *et al.*, 2011; Johannessen *et al.*, 2005). Pathogen in soil can spread by runoff and leaching and contaminates neighboring soil and water bodies (Jablasone *et al.*, 2004; Santamaría & Toranzos, 2003). The use of contaminated irrigation water can be an important factor in vegetable contamination (Oliveira *et al.*, 2010). The pathogen in runoff and leachate can cycle back to the farm or even affect neighboring farms if the contaminated water is used for irrigation purpose (Bourn & Prescott, 2002). The vegetables could easily get in contact with the soil even after harvest which may be contaminated if the soil contains pathogen (Santamaría & Toranzos, 2003). Several farm practices increased the risks of *E.coli* contamination.

These could be due to:

- use of animal manure with ageing period less than 6 months from fertilization to harvest,
- manure application within the interval of less than 90 days before harvest,
- use of non-composted manure,

- use of cattle manure which has higher chances of *E.coli* contamination compared to other types of manure, and
- use of contaminated water for irrigation (Mukherjee *et al.*, 2007; Oliveira *et al.*, 2010).

Humans who consumed the contaminated vegetables can get infected which leads to food poisoning. The food poisoning incident had raised the concern of food production safety. The certified organic farms are restricted to apply untreated manure less than 90 or 120 days before crop harvesting depending on the contact level between edible part and soil (USDA, 2011; Winter & Davis, 2006). Currently, the FDA is finalizing the Rule for Produce Safety which is under the Food Safety Modernization Act (FSMA). The regulation proposed to extend the waiting period of between the application of raw manure and crop harvesting. In addition, the removal of the proposed 45 day minimum application interval for compost will encourage farmers to compost manure (FDA, 2013). Food handling at postharvest stage (transportation, processing and packaging) is also one potential pathway for contamination to take place. The CODEX General Principles of Food Hygiene and Hazard Analysis and Critical Control Point (HACCP) system are some of the guideline and safety programme available to ensure safety of food production (FAO, 2000).

### **2.8.9 Soil degradation**

Soil degradation is the decrease in soil fertility and productivity due to improper land use which is a major threat to agricultural sustainability and environmental quality (Lal, 1993). Most of the world's agricultural soils have become depleted in organic matter and soil health over the years under intensive agriculture, compared with their state under natural vegetation (Table 2.15) (Corsi *et al.*, 2012). Almost 6–40% of global

terrestrial area is lost due to soil degradation (Horrigan *et al.*, 2002; Tschardtke *et al.*, 2012). It is estimated that worldwide land degradation has led to 20–50,000 km<sup>2</sup> of soil lost annually (Montanarella & Vargas, 2012). In the historic past, about one-third of the agriculture land has been affected by soil degradation (Hurni *et al.*, 2008). The developing region of Asia (mostly South Asia) and Africa (mostly sub-Saharan Africa) are forecasted to face issues with limited land and soil degradation because of high population growth (Lal, 2009). The loss of nutrients alone resulting from soil erosion has an estimated cost to the United States of up to \$20 billion a year (O’geen & Schwankl, 2006; Tegtmeier & Duffy, 2004).

Table 2.15: Global degraded dry land (million ha)

Area	GLASOD <sup>a</sup>
Africa	321
Asia	453
Australia and Pacific	6
Europe	158
North America	140
South America	139
World (Total)	1216

Source: Gibbs & Salmon (2015)

<sup>a</sup> Global Assessment of Soil Degradation

Soil degradation occurs when the soil lost exceeds the natural regeneration rate (Papendick & Parr, 1992). Improper farm management such as over ploughing, inadequate water use, drainage and inadequate plant residues management often leads to soil salinization, erosion, and desertification which are the major cause of soil degradation (Abd-Elmabod *et al.*, 2012; Gretton & Salma, 1996; Khresat, 2014; Snakin *et al.*, 1996).

Generally, soil degradation is divided into physical, chemical and biological degradation (Imeson, 1995; O'geen & Schwankl, 2006; Oldeman, 1994; Snakin *et al.*, 1996):

- i. Physical: compaction, soil erosion by water forced or wind forces, reduce soil permeability
- ii. Chemical: loss of organic matter, nutrients and fertility, salinization, acidification, accumulation of chemical and toxic substance from agrochemical
- iii. Biological: imbalance soil biodiversity and increase pathogen population.

It is estimated that 655 million hectares agriculture land are affected by soil degradation where 55.6%, 27.9%, 12.2% and 4.2% were due to water erosion, wind erosion, chemical and physical degradation, respectively (Hurni *et al.*, 2008) (Figure 2.12). Thus, the major drivers for soil degradation in agriculture land are water erosion, wind erosion, tillage erosion, crop harvesting, topography, rapid loss of organic matter, organic carbon and nutrient, lack of vegetation cover, inappropriate farm practices, land marginalisation (SoCo, 2009). Soil erosion by intensive irrigation is one of the major culprit in soil degradation in agriculture land (Chao-Yin *et al.*, 2012; Oldeman, 1994).

Even though soil erosion is a natural process, the bare soil and over tilled agriculture soil has accelerated the rate of erosion. Rainfall and irrigation hit on exposed soil caused soil particle to detach from the aggregate soil and when runoff rate is higher than water infiltrates the soil, the runoff will wash away the soil particle (Ahn *et al.*, 2013; Pimentel *et al.*, 1995). In addition, small and light weight soil particle is transported away from agriculture land by wind, especially at steep cropland. One of the key indicator of erosion by water is the suspended solid of runoff from agriculture land (Bechmann *et al.*, 2008).

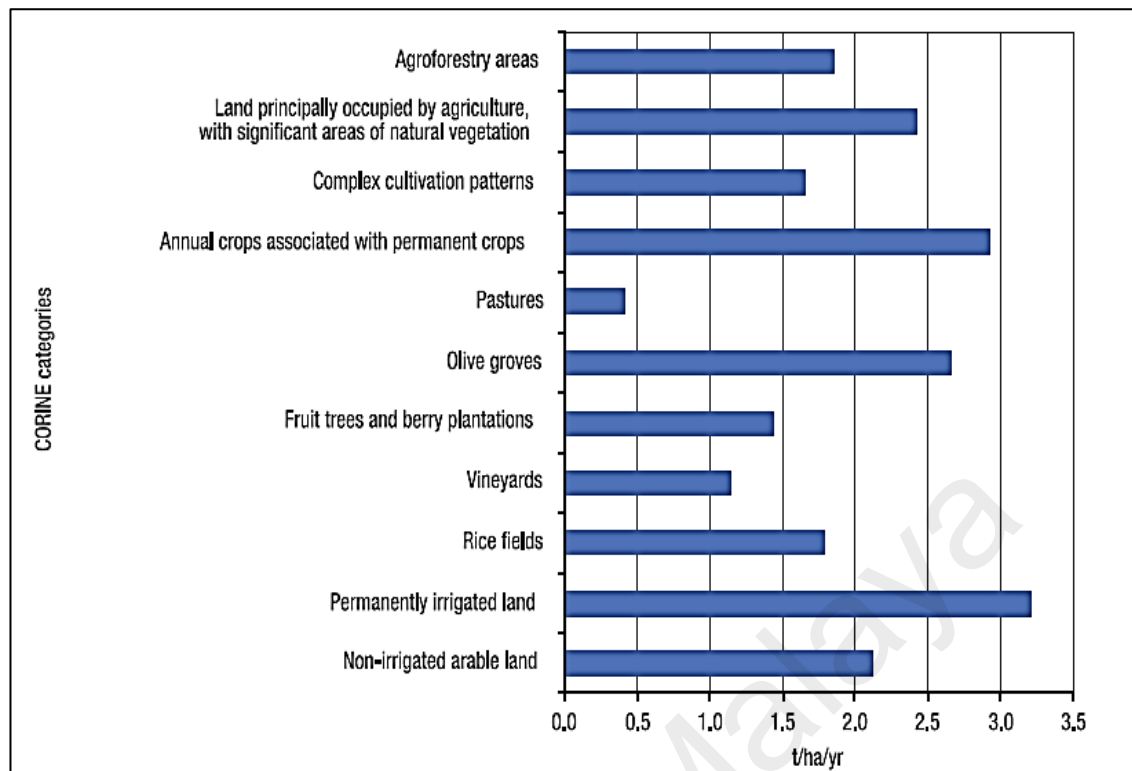


Figure 2.12: Erosion risk of various farming systems (SoCo, 2009)

Soil productivity is vital for food production thus it is crucial for soil conservation and preservation. The loss of soil productivity can directly suppress crop growth which affects the production efficiency (Lal, 2009). Thus, it is important to prevent soil degradation instead of attempting to cure a degraded soil (Wu & Ma, 2015). In addition, the cost of conservation is much cheaper than replenish the lost nutrient back into soil. It is estimated that every USD1 spent on soil conservation will be able to save USD 5 on replacing nutrient and water lost due to soil erosion damage (Pimentel *et al.*, 1995). The key to maintain soil quality is ensuring regular input of organic matters such as crop residues, livestock manures, and compost. In addition, farm practices such as crop rotation and reduced tillage will assist in reducing soil organic matter lost (Dalzell *et al.*, 2013). Some of the proven conservation practices are no-till cultivation, strip cropping,



ridge-planting, grass strips, mulches, living mulches, agroforestry, terracing, contour planting, cover crop and windbreaks (SoCo, 2009).

## **2.9 Sustainable agriculture**

The world is facing unprecedented challenges in combating world hunger, fulfil world food demand and at the same time dealing with extreme weather events caused by climate change (Beddington *et al.*, 2012a). In order to meet the increased world population, there is a need to increase food production to meet the increased population (FAO, 2015c). There is a close linkage between agriculture growth and the eradication of poverty and hunger. According to FAO, agriculture GDP growth is more effective in reducing world hunger and poverty than non-agriculture sectors (FAO, 2015c). The agriculture expansion is crucial in combating poverty and hunger problems. However, conventional and industrial agriculture often associated with farm practices such as monocultures, excessive use of synthetic chemical pesticides and fertilizers, and unsustainable water consumption are detrimental to the environment (Horrigan *et al.*, 2002). All these lead to environmental pollution, loss of biodiversity, soil degradation, and wastage of resources. The question is can sustainable agriculture be achieved with a rapidly degrading natural resource base, increased use of fertilizers, pesticide, and fresh water combined with increased land expansion (Wu & Ma, 2015)?

Increased crop production and productivity should not come at the expense of the environment (Chen, 2011). Thus, it is necessary to adopt sustainable agricultural development such as resource conservation, environmental impact mitigation, global climate change mitigation and adaptation, which are integral to any agricultural program aimed to increase production. The agenda of Post-2015 and The Sustainable Development Goals have identified sustainable agriculture as the principle strategies development approach (FAO, 2015c). In 1987, the term “sustainable development” was

first defined by the Brundtland Commission (formally known as World Commission on Environment and Development) by the United Nations (Lichtfouse *et al.*, 2009, p. 3; Pretty, 1995):

“Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs.”

The first definition of sustainable agriculture used by United States was defined by the American Society of Agronomy (Siwar & Hossain, 2001, p. 33):

“A sustainable agriculture is one that, over the long term, enhances environmental quality and the resource base on which agriculture depends; provides for basic human food and fibre needs; is economically viable; and enhances the quality of life for farmers and society as a whole.”

The definition of sustainable agriculture was then endorsed via United States Farm Bill under the Food, Agriculture, Conservation, and Trade Act of 1990 (FACTA), Public Law 101-624, Title XVI, Subtitle A, Section 1603 (Gold, 2007, p. 4; Siwar & Hossain, 2001, p. 33). The term sustainable is defined as:

“An integrated system of plant and animal production practices having a site-specific application that will, over the long term: satisfy human food and fibre needs, enhance environmental quality and the natural resource base upon which the agricultural economy depends, make the most efficient use of non-renewable resources and on-farm resources and integrate, where appropriate, natural biological cycles and controls, sustain the economic viability of farm operations and enhance the quality of life for farmers and society as a whole.”

It is impossible to have a precise and absolute definition of sustainable agriculture as it is a complex and contested concept (Pretty, 1995). A sustainable farm system should ensure balance between environment, economic and social benefit to farmers by ensuring efficient production while conserving resources and the environment (Duesterhaus, 1990). Sustainable agriculture aims to achieve sustainable profit over the long period of time under the stewardship of air and water which ensured quality of life for farmer communities (SARE, 2010). Thus, profit oriented farming system that neglects the environmental quality is considered unsustainable.

Increased food demand and the concern on climate change has raised the interest of scientist, environmentalist, activist and industry player on sustainable agriculture (Kleijn *et al.*, 2009). The Commission on Sustainable Agriculture and Climate Change reported that sustainable agriculture is the solution to poverty and will be able to combat climate change impact on agriculture which could ensure food security, improve farmers income, reduce environmental impact and at the same time adapting to climate changes (Beddington *et al.*, 2012b). Proper farm practices can increase crop productivity but also simultaneously improved soil organic carbon, soil microbial biomass, soil microbial activity which then could preserve soil resources and fertility.

The USEPA has identified several farm practices such as organic farming, Best Management Practices (BMP), agro-ecological practices and Integrated Pest Management (IPM) practices as some of the sustainable agriculture approach (Table 2.16) (Cestti *et al.*, 2003; Hartz, 2006; Lichtfouse *et al.*, 2009; Wezel *et al.*, 2014).

Table 2.16: Farm practices that function to support sustainable agriculture

Practices	Relevant management						
	Soil	Nutrient	Pesticide/ Herbicide	Water	Air (GHG emission)	Biodiversity	Cost and profit
Incorporation of organic materials	X	X			X		
Cover crop and vegetation barrier	X	X		X		X	
Contour, terrace farming	X	X	X	X			
Strip, mixed cropping	X	X	X	X			
Fallow management	X	X	X		X		
Diversion	X						
Crop rotation management	X	X			X		
Rainwater harvesting	X	X		X			X
Precision irrigation	X	X		X			
Water way protection system, Subsurface drainage	X			X			
Runoff and groundwater monitoring	X	X	X	X			
Conservation tillage	X	X			X	X	
Sediment retention and erosion control	X			X			
Precision nutrient management		X		X	X		
Precision pesticide and herbicide management			X	X			
Spillage prevention	X	X	X	X	X		
Seed bed prepared right before planting	X			X	X		X
Manure management system, eg. digestion before application		X		X	X		X
Selective timber harvest	X	X			X	X	
Tree planting	X	X			X	X	
Ensure productivity and sustainability							X
Identify farming practices and technologies that balance the nutrients	X	X	X	X	X	X	X
Public education	X	X	X	X	X	X	X

Source: Cestti et al., 2003; Wolff, 2003; Wu & Ma, 2015

Various innovative good farm management methods have been developed to achieve sustainable agriculture (Goulding *et al.*, 2008; Horrigan *et al.*, 2002; Lal, 2009; SARE, 2010; Wezel *et al.*, 2014; Wu & Ma, 2015) and these are:

1. Ecological pest and weed management which used a combination of biological, physical, or chemical strategies such as trap crops for insect pests, physical removal of weeds and insects, natural pesticides, selecting crops that suppress weeds growth and creating environment for beneficial insects or application of chemicals if necessary.
2. Grazing management such as rotational system ensures animals moving from pasture to pasture.
3. Conservation tillage such as contour tillage, reduced tillage and no-till able to prevent and reduce soil loss due to wind erosion, water erosion, reduce soil compaction, conserve water and carbon sequestration.
4. Cover crops such as rye, clover or vetch cultivation after harvest can suppress weed and insect growth, reduce erosion control and improve soil quality (Gray & Trigiano, 2014).
5. Increased farm biodiversity by growing variety of crops such as intercropping, relay intercropping and agroforestry with timber, fruit, or nut trees. This allows farm to be more diseases and pests resilient, conserve soil and water, provide suitable growing environment for flora and fauna, increase beneficial insect populaces, adaptable to extremes weather and market conditions (Dordas, 2009).
6. Good nutrient management will be able to improve soil fertility, decrease the incidence of diseases in crops, reduce or even prevent water pollution and reduce operation costs due to less purchases of fertilizer. The use of organic fertilisation, compost, vermicompost, split fertilisation and biofertilizer can

improve soil structure and reduce nutrient loss (Masso *et al.*, 2015; Sinha *et al.*, 2014).

7. Soil management is crucial to ensure the soil fertility. The use of biosolids and crop residues increase soil organic matter that enhance microbial activity and restore degraded soil, minimum soil disturbance (cover crops and mulching) reduce erosion, sub-soil fertigation to ensure adequate amount of water for crop growth and minimize leaching and runoff, diversify cropping/farming systems which support nutrient cycling and nutrient use efficiency .
8. Implement variety of marketing techniques such as value-added products, branding, understand and meeting market demand via market research; direct selling to customer and or restaurants via delivery, farmers market, roadside stands or internet.
9. Integrated farm management which combines the various practices to achieve balance between farming and nature which ensure farm profitability by efficient use of resources (Wu & Ma, 2015).

Such practices encouraged farmers to focus on long-term planning and have greater consideration for environmental impacts, rather than only focusing on yield-scaled profit (Wu & Ma, 2015). Because each farming system is unique in its own way and is environmental sensitive, farmers have to design their own innovative agricultural systems which are suitable for the farm sustainable development (Lichtfouse *et al.*, 2009; Pretty, 1995).

Malaysian government realised the need for sustainable agriculture especially for long term benefit. Thus, together with the national policy like the Third National Agriculture Policy (NAP3) it has initiated the agriculture transformation plan to ensure sustainability of agriculture sector (Ahmad, 2001). In the effort to promote sustainable

agricultural practices, Malaysian Department of Agriculture (DOA) has initiated programmes that provide farming training and certification such as Malaysia Good Agriculture Practice (MyGAP) and Malaysian Organic Certification Program (SOM) (Tiraieyari *et al.*, 2014). In 1996, a survey showed the Malaysian farmers' awareness level on environmental impact to agriculture activities was generally moderate to high (Barrow *et al.*, 2005; Barrow *et al.*, 2009; Barrow *et al.*, 2010). However, the practice of conservational farming method was low as the farmers have higher priorities on higher economic return rather than environmental concerns (Midmore *et al.*, 1996). With the effort done by the government, the awareness of sustainable agriculture has increased tremendously which in year 2014 there is a total of 3,200 MyGAP certified farms and 49 SOM certified farms (MPC, 2014; PEMANDU & EPU, 2014).

## **2.10 Nutrient Management**

Nutrient management is a systematic management by farmers to manage the quantity, source, placement method, and timing of the application of nutrients to plants (Chopra *et al.*, 2014). Nutrient management is identified as one of the potential tools in agricultural economy and social sustainability by improving efficiencies of nutrient use for farm system (Goulding *et al.*, 2008; Wezel *et al.*, 2014). Good nutrient management will be able to improve soil fertility, decrease the incidence of diseases in crops, reduce non-point source pollution, lower greenhouse gas emission, optimize fertilizer usage and productivity which are able to reduce farm operational costs (DEFRA, 2015b; Dordas, 2009; Wezel *et al.*, 2014). Competent use of fertilizer is able to reduce the input costs and environment conservation costs. The key in nutrient management is to identify potential improvement of current farm nutrient regime employed by farmers that could enhance crop yield and reduce impacts on water quality (Cestti *et al.*, 2003; Defoer *et al.*, 1998). This can be achieved by understanding soil nutrient status, crop nutrient

requirements and nutrient flow monitoring (Bechmann *et al.*, 2008). Nutrient management should be site specific as each individual farm is different in terms of management, topography, and soil texture (Cestti *et al.*, 2003). Efficiency in nutrient use can be increased by;

1. Appropriate fertilizer application timing,
2. Systematic nutrient accounting,
3. Assessment of nutrient cycling (understanding of complex interrelation of plant, soil and water), and
4. Utilization of organic nutrient, symbiotic microbes, and legume (Dungait *et al.*, 2012).

Monitoring, spacial measurement and modelling approach are important in nutrient management especially when formulating a nutrient management regime. The competent use of fertilizer can be achieved by good understanding of nutrient flow within the farm system. Through nutrient budgeting and nutrient balance the nutrient flux through various key processes are quantified and traced (Dungait *et al.*, 2012). It is a difficult and complex task to match the supply of soil nutrient to crop demand due to variations and dynamic interactions of nutrients in terms of physical, chemical, biological and climate in each farming site (Goulding *et al.*, 2008). Thus, site specific study of nutrient management is important, as it is reported that site specific nutrient management increases nutrient use efficiencies on an average of 30-40% in paddy fields (Dobermann & Cassman, 2002; Dobermann *et al.*, 2002). Nutrient management could be developed by adopting the following steps (Chopra *et al.*, 2014; Laboski & Peters, 2012; NRCS, 2012):

1. Nutrient balance and budget are established by considering all potential nutrient input-output such as fertilizer, soil amendments, compost, crop residues,



irrigation water, rainfall, soil biological activity, animal manure, leaching and runoff (Oenema *et al.*, 2003).

2. Nutrient analysis of input-output are determined prior to application and all input that contributes to nutrient input must be accounted for in the nutrient budget (Rosen & Eliason, 1996).
3. Determine suitable fertilizer application. There are several methods of fertilizer application; examples are broadcasting, point placement, band placement, fertigation: drip irrigation, root dipping, and foliar spray. Selection of the method varies from farm to farm and it depends on the availability of labour, investment, crop system, crops type, soil properties and etc (Yinbo *et al.*, 1997).
4. Nutrient application rate, nutrient source, application timing, nutrient placement are crucial in minimizing environmental risk and nutrient loss via leaching and runoff (NRCS, 2012). All these are highly dependent on production goal, cropping pattern, soil properties, soil organic content, soil salinity, soil pH, soil biological activity, tillage, crops types, plant nutrient uptake, cropping system, rotation, climate, drainage condition, and risk of nutrient loss.

Several strategies are available to be incorporated into the nutrient management regime (Goulding *et al.*, 2008; Hochmuth, 2003; NRCS, 2012; Simonne & Hochmuth, 2005):

- Reduce tillage, no till, or strip till increases soil organic matter and soil aggregation stability, reduce soil compaction, improve water infiltration rate, biological fixation and nutrient use efficiency.
- Use of cover crop and crop rotations to improve nutrient cycling. Cover crops between cropping season to increase soil organic matter, nutrients sequestration, minimize nutrient loss, reduce soil erosion by wind and rainfall, and enhance nutrient cycling.

- Double cropping increases nutrient use efficiency and reduce nutrient loss
- Precision farming where nutrient application rate should be site specific based on the crop yield, soil properties, soil nutrient content and plant nutrient analysis, e.g. Pre-side-dress soil nitrate testing (PSNT) is useful in nutrient management by estimating soil N availability in which additional N fertilization can be delayed or reduced (Hartz, 2006).
- Nutrient management monitoring via analysis of crop nutrient content to monitor
- Yield monitoring of each plot and making nutrient input adjustment accordingly
- Mulching to control nutrient losses, e.g. plastic mulch and strip mulch
- pH monitoring to maintain the soil pH in a range that ensured optimal crop nutrient utilization and adequate nutrient availability, e.g. high pH resulting binding of micronutrients to soil particle which are not accessible by crops.
- Integrate organic input such as livestock manure, compost and crop residues into the nutrient program. The organic material supplementing nutrient and in the same time increases soil organic matter which improves soil structure.
- The use of controlled release fertilizer such as polymer coated urea, sulphur coated urea, and isobutylene di-urea. The slow nutrient releases fertilizer and reduces the risk of losses.
- Water management and nutrient management are interrelated. Applications of irrigation water could minimize the risk of nutrient loss to surface and groundwater.
- Fertigation and drip irrigation supply nutrient according to plant requirement to reduce nutrient loss and increase water use efficiency, e.g. drip irrigation improves water use efficacies by reducing 50% to 70% of water usage (Hartz, 1996), and

- Appropriate soil preparation before planting is crucial for uniform growth of crops. Deep plough of cover crop and crop residues 6 to 8 weeks prior to planting allows the decomposition process to breakdown the organic matter, increase nutrient availability during crop planting stage and reduce the risk of damping off diseases.

The basic principle of integrated nutrient management (INM) is judicious, efficient and integrated use of various nutrient sources such as organic fertilisation, microbial inoculant, compost, vermicompost, crop residues, chemical fertilizer, biological fixation and biofertilizer with innovative technologies (nitrification inhibitor and controlled release fertilizer) to ensure sustainable production (Hochmuth, 2003; Khan *et al.*, 2008; Noor *et al.*, 2008; Shukla *et al.*, 2014). The INM has been proposed as a tool for soil conservation and enhanced soil fertility. Research shows that the integrated use of organic matter and chemical fertilizer can reduce soil fertility depletion, reduce production costs, increase yield and in the same time reduce the use of synthetic fertilizer (Khan *et al.*, 2008).

### **2.11 Agri-environment indicators**

Agri-environmental indicators provide insight of the agriculture activities impact on the environment (Halberg *et al.*, 2005; Niemeijer & de Groot, 2008). It comprises of physical, biological and chemical indicators that are specific to environmental pressures, conditions and responses. Agri-environmental indicators are regularly used to evaluate the agricultural sustainability and farm level indicator allows farmer to self-assess the sustainability of their farm (Bélanger *et al.*, 2012). Agriculture is a complex system, therefore farm management is an on-going process that requires constant monitoring and decision making both short-term and long-term to adapt to the changes affected by

economic, environment, political, social, and climate (Smit & Skinner, 2002; Yli-Viikari *et al.*, 2007).

Farm management and environment has direct and indirect linkage to the farm emission that impacts the environment (van der Werf & Petit, 2002). The decision making by government, agri-business and individual farmers involved in agriculture require systematic indicators to evaluate the consequences of the decisions made on the environment (Hřebíček *et al.*, 2013; Moxey *et al.*, 1998; Oñate *et al.*, 2000; Vizzari *et al.*, 2015). The indicators act as a decision supporting tool to provide valuable information on how agriculture changes affect soil, water and environmental quality (Eilers *et al.*, 2010). In addition, the indicators assist farmers to achieve sustainable, quality, competitive and environment sound farming practices (Montero *et al.*, 2007).

The increased concerns on agricultural impact on environment have led to the development of agri-environment assessment tool with the objective of achieving sustainable agriculture. In 2013, the OECD, in conjunction with European Union Statistical Office (Eurostat) and the FAO have published the OECD Compendium of Agri-Environmental Indicators to measure the environmental performance in agriculture (Hsu *et al.*, 2014; OECD, 2013). The IRENA operation (Indicator Reporting on the Integration of Environmental Concerns into Agriculture Policy) by EU has identified 35 agri-environmental indicators for monitoring the environmental performance of agriculture (EUROSTAT, 2015). The UK Department for Environment, Food & Rural Affairs (DEFRA) has identified 10 indicators of greenhouse gas emissions from agriculture (DEFRA, 2014d). In addition, other institutions, organizations and NGOs like European Economic Area (EEA) and FAO have also developed sets of indicators guidelines (Hřebíček *et al.*, 2013). Generally, agriculture environmental indicators cover

eight domain: Air & Climate Change, Energy, Fertilizers Consumption, Land, Livestock, Pesticides, Soil, and Water (FAO, 2014a). Agri-environment indicator is an alternative farm system assessment that can be used as a benchmark for farm management decision making. There are two types of agri-environment indicators (Figure 2.13) (Oñate *et al.*, 2000; van der Werf & Petit, 2002):

- i. Means-based indicators are the farm practices that may cause emission, e.g. excessive fertilizer application, and
- ii. Effect-based indicators are monitoring of the emission, state or impact of the farming system that are caused by farm practices (cause-effect relations), e.g. nitrate concentration in water (Bockstaller *et al.*, 2015).

Mean-based indicator is a straight forward and less costly method which highlights changes in management or environmental sensitivity (Bélanger *et al.*, 2012; Willenbockel, 2014). Research showed that higher farm input do not indicate higher environment pressure (Gaudino *et al.*, 2014). Thus, mean-based indicator may not be able to reflect the real situation on environmental impact. On the other hand, effect-based approaches were able to deliver better outcomes by allowing integration of existing knowledge with innovative management, reduce managerial restrictions and regulations (Burton & Schwarz, 2013; Qiu *et al.*, 2010). The disadvantage of effect-based indicator is the inability to identify the weaknesses of the current farm management and how to improve it (Moxey & White, 2014). In addition, it is extremely costly for farmer to analyse the water and soil samples regularly (Matzdorf & Lorenz, 2010; Oñate *et al.*, 2000). Therefore, the combination of mean-based and effect based indicators are able to provide a more comprehensive assessment. From the mean-based and effect based indicators, it can be further divided into state indicators (current situation), risk indicators (environment impact caused by agriculture practices),

response indicators (management or practices that results in environment disturbance) and efficiency indicators (resource use efficiency) which are based on the Driving forces-Pressures-Impacts-Responses (DPIR) framework (Massé *et al.*, 2013; Niemeijer & de Groot, 2008; Zalidis *et al.*, 2004).

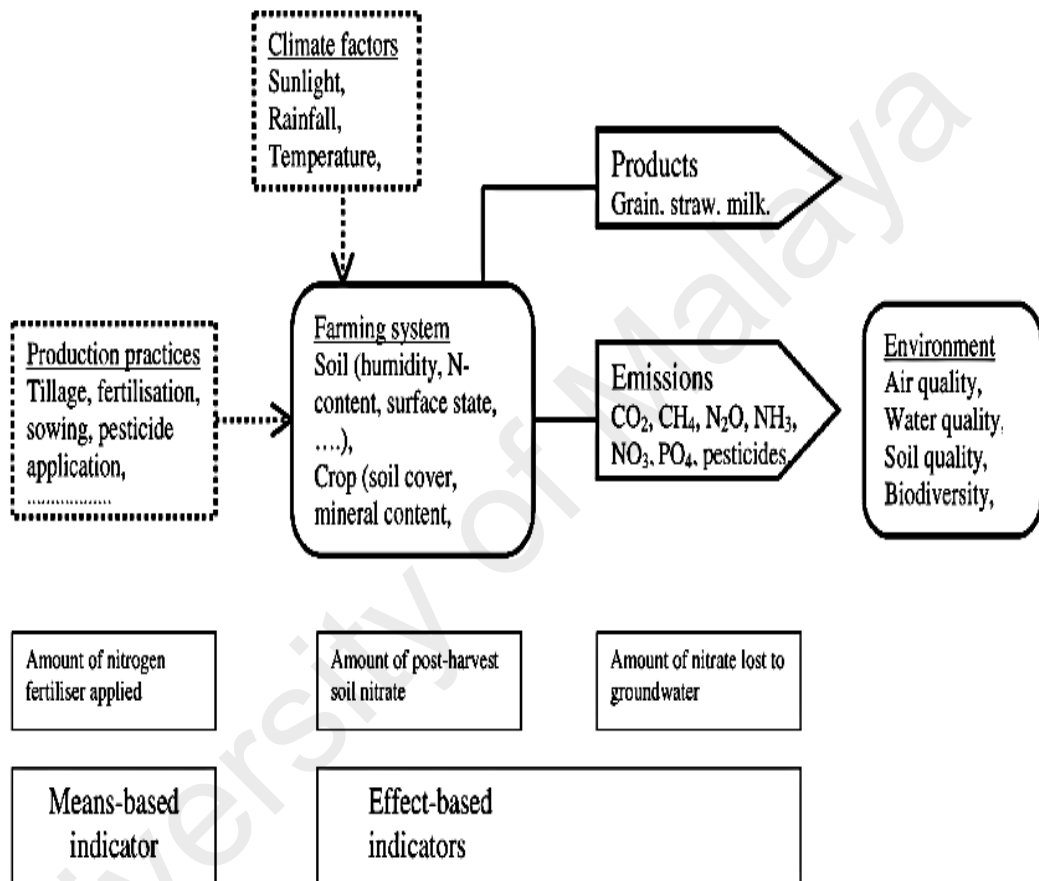


Figure 2.13: Factors affecting mass flux and emissions in a farming system (van der Werf & Petit, 2002)

There are several agri-environment indicator assessment tools used in Europe such as: REPRO, DLG, KUL/USL, KSNL, Indigo, SALCAKUL/USL, (Bockstaller *et al.*, 2009; Hřebíček *et al.*, 2013). A basic six steps framework was suggested by Bélanger *et al.* (2012) to develop the farm assessment tool. The definition of sustainability varies

between region and country, thus the first thing is to develop a farm level assessment tool in defining the environmental sustainability concept. The second step is determining the objective and principle of the assessment. Third, is identifying and selecting suitable indicators. Fourth, is establishing the baseline value for references. Fifth, is evaluating the selected indicator whether it is suitable to achieve objective and lastly is selecting the ultimate indicators set (Niemeijer & de Groot, 2008). The assessment should utilize simple indicator calculation to allow feasibility of assessment and reduce the risk of errors (Hřebíček *et al.*, 2013). The criteria used in indicator selection are (Montero *et al.*, 2007; Schröder *et al.*, 2004):

1. Objective oriented with clear and unambiguous definition,
2. Convenient, integrated and analytically sound,
3. Indicator should be responsive, immediate and straightforward for individual to interpret,
4. Cost efficient and sufficiently accurate, and
5. Relevant to current issue and policy which is sensitive to changes within policy time-frames (Niemeijer & de Groot, 2008).

Agri-environment indicators provide benefits such as (Piorr, 2003):

- Provide relevant information on the current state of agriculture environment to decision makers and general public, e.g. agri-environmental reports (Lefebvre *et al.*, 2005),
- Provide an overview of the relationship between farm practices, agriculture policy and environment to decision maker, e.g. international or national development plans and strategies,
- Constant monitoring of the effectiveness of the implemented sustainable agriculture measures. It provide feedback and evaluation on regulations,

conventions, environmental initiatives, and progress of achieving environmental goals e.g. documentation of farming practices, weather conditions, runoff and water quality (Bechmann *et al.*, 2008),

- Enable a quantitative and qualitative environmental observation, and
- Allow comparison of different farm practices or farming systems for research purposes (Langeveld *et al.*, 2007).

Limitations of agri-environment indicator assessment are:

- i. Constrained by data availability and coverage (Moxey *et al.*, 1998),
- ii. Reflect only the condition within farm system at a particular temporal and spatial space (Langeveld *et al.*, 2007),
- iii. Mean-based data is not easily available and analysis can be costly. In addition, the quality of the data may affect the assessment (Yli-Viikari *et al.*, 2007), and
- iv. Data required reference value for interpretation. The reference value can be regulation threshold, a norm, a target, research results or relevant statistics (Hřebíček *et al.*, 2013).

This study utilizes several agri-environment indicators to assess the organic and conventional farm system as in Table 2.17.



Table 2.17: Agri-environmental indicators assessment used in this study

Component	Indicators	Principle	Types of indicator	References
Farm material	‘Flow’ indicators	Evaluate the management of material input-output flows within a system which able to describe the structure and functions of the system.	Mean based	(Yli-Viikari <i>et al.</i> , 2007)
Fertilizer management	Nitrogen balance /Nitrogen budgets/ Nitrogen accounting	Measure the total N inputs into defined system (e.g. farm-gate, soil surface or soil systems) and subtracts quantities total N outputs from the system.	Mean based	(Buckley & Carney, 2013; Rozemeijer <i>et al.</i> , 2014; Schröder <i>et al.</i> , 2004)
	Nutrient Surplus	Defined as the difference between total N input and total N export through crop harvesting.	Mean based	(Langeveld <i>et al.</i> , 2007)
	Fertilizer use efficiency	Defined by the fertilizer use per crop production	Mean based	(Hřebíček <i>et al.</i> , 2013)
Carbon sequestration	Carbon balance / Carbon budgets/ Carbon accounting systems	Measure the carbon inputs onto the system and subtracts quantities exported from the farm through outputs and evaluate the input-output balance	Mean based	(Hřebíček <i>et al.</i> , 2013; Langeveld <i>et al.</i> , 2007)
Organic matter	Biomass balance	Difference between input of organic matter (in fertilizers and plant residues) and crop harvesting	Mean based	(Hřebíček <i>et al.</i> , 2013)
Soil	Soil carbon and nitrogen content	Soil carbon, nitrogen and nitrate levels based on legislative standards or recommended practices in literature	Effect-based	(Bélanger <i>et al.</i> , 2012; Langeveld <i>et al.</i> , 2007)
Water	Runoff and leachate chemical composition	Selected chemical levels based on legislative standards or recommended practices in literature	Effect-based	(Salleh & Harun, 2014)
	Integrated water management	Evaluate water use efficiency by calculation of water use per production	Mean based	
Air	GHG emission	Measured of CO <sub>2</sub> , CO, NH <sub>3</sub> , N <sub>2</sub> O, and CH <sub>4</sub> emissions	Effect-based	(Yli-Viikari <i>et al.</i> , 2007)
Production	Production efficiency	Calculate by production per area of land use	Mean based	(Hřebíček <i>et al.</i> , 2013)

## 2.12 Nutrient balance

Researchers have emphasized the importance of improving nutrient management, reducing nutrient losses, and recycling nutrients (Sutton & Bleeker, 2013; Zhao *et al.*, 2010). A detailed and quantitative understanding of nutrient balance in various farming systems is a prerequisite for achieving proper nutrient management. Nutrient balance assessment is based on static modelling systems of input-output of a farm system (Figure 2.14) (Roy *et al.*, 2003). Nutrient balance is the most common agri-environment indicator which is based on the principle of budgeting the farm's input-outputs to calculate the balances (Hřebíček *et al.*, 2013). It is also the basic principle of integrated nutrient management to budget the farm input-output (Smaling *et al.*, 1993; Wu & Ma, 2015).

Soil nutrient balance is the difference between nutrient inputs (fertilizer, manure, and compost) and nutrient outputs (harvested crops, leaching, gaseous emission) (Bindraban *et al.*, 2000). Soil nutrient balance indicates the annual nutrient loadings of substances such as nitrogen or carbon input to agricultural soils (DEFRA, 2014b). It is also the indicator of environment pressure from agriculture activities (DEFRA, 2015a). A positive nutrient balance indicates a potential loss of nutrients to the environment or nutrient accumulation in soil, whereas a negative balance signifies soil nutrient depletion (Bouwman *et al.*, 2013). Nutrient balance can be used to manage a variety of substances or elements, and N, P, and C are the most common elements being monitored in agriculture soil due to its high impact on environment and its importance in sustainability agriculture (Dungait *et al.*, 2012).

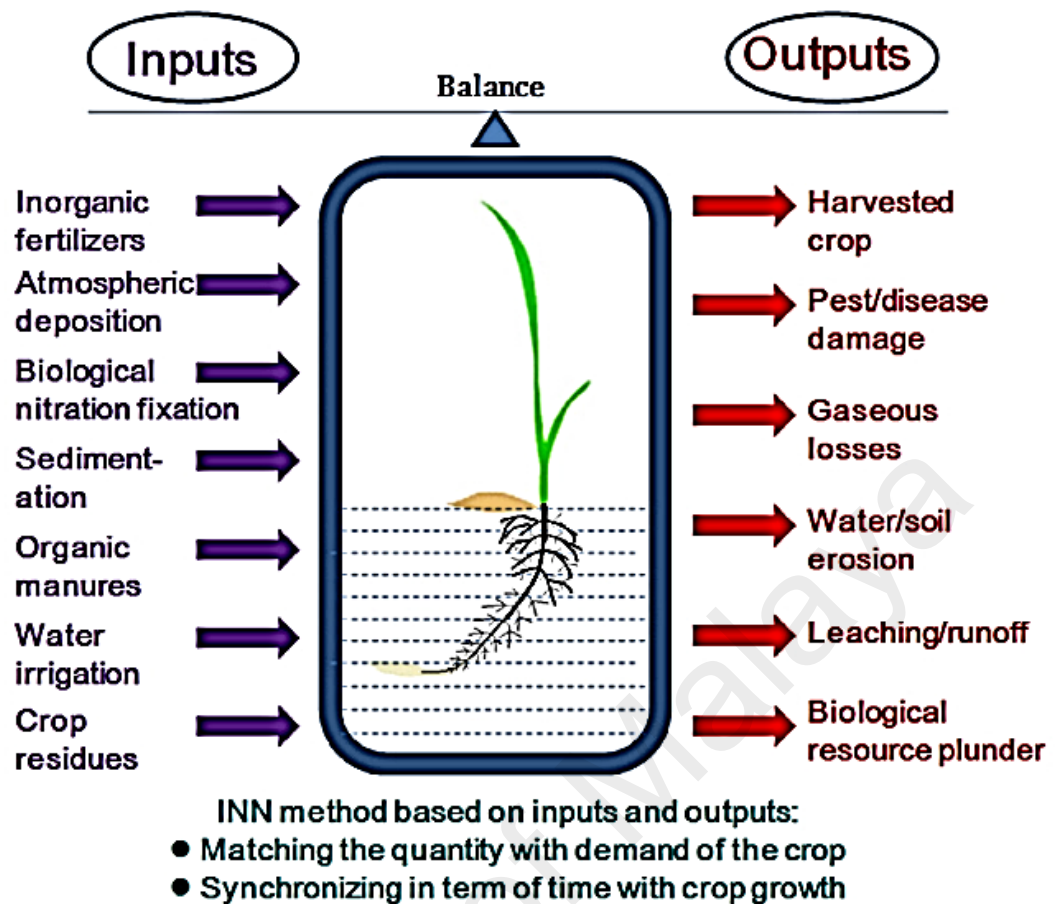


Figure 2.14: Nutrient balance/nutrient budget/nutrient accounting is the basic principle of integrated nutrient management (Wu & Ma, 2015)

Nutrient balance is applied at various scales from plot and catchment to regional and global assessment (Scoones & Toulmin, 1998). Detailed scale nutrient balance such as farm level nutrient balance plays an important role on the sustainability of agricultural production systems (Bindraban *et al.*, 2000; Oenema *et al.*, 2003). Nutrient balance is significantly influenced by farm practices (Wortmann & Kaizzi, 1998). Thus, the assessment of farm-level nutrient balance provides an overview of current farm practices and offers information that can be used to improve nutrient use efficiencies by (DEFRA, 2014b):

- i. Increase the understanding of nutrient cycling,

- ii. As performance indicator and awareness raiser in nutrient management and environmental policy, and
- iii. As regulating policy instrument to enforce a certain nutrient management policy in practice.

Generally three categories of nutrient balance are available (Brouwer, 1998; Oenema *et al.*, 2003):

- i. Farm-gate: integrated measurement of environmental pressure and suitable as environmental performance indicator,
- ii. Soil surface: estimating the nutrient loading of the soil, and
- iii. Soil system: detailed budget of nutrient inputs and outputs, nutrients cycling within the system, nutrient loss pathways and changes in soil nutrient pools.

Challenges for nutrient balance analyses are (Roy *et al.*, 2003; Scoones & Toulmin, 1998):

- Difficulty to extrapolate the model due to non-linear data and micro-level diversity,
- Provide only a snap-shot view that unable to capture soil dynamic processes,
- Accuracy of the nutrient balance is highly variable due to biases, errors, and data collection, e.g. personal bias, sampling bias, measurement bias, data manipulation bias and fraud (Oenema *et al.*, 2003), and
- Lack of integration of socio-economic issues that influence farmer's nutrient management.

It must be recognised that nutrient balance is not a definitive statement; it is an indicator for further discussion or research development. Nonetheless, with appropriate

methodological refinements, effective sampling strategy, increase spatial explicit data, assumptions and uncertainties are made explicit and proper calculation and quantification of flow will minimise the methodological gap (Roy *et al.*, 2003; Scoones & Toulmin, 1998).

### 2.13 Nitrogen balance

Nitrogen is one important element in agriculture as it is often the limiting factor of crop growth (Rosen & Eliason, 1996). Addition of N fertilizer by farmer is required to achieve target yield and a survey showed 50% of the farmers apply excessive fertilizer to ensure yields (Arbuckle & Rosman, 2014; Jarvis *et al.*, 2011). However, several researches show that only 20% to 68% of N applied in the field is taken up by crop (Table 2.18) (Dungait *et al.*, 2012; Fortes *et al.*, 2011; Gardner & Drinkwater, 2009; Goulding *et al.*, 2008; Yan *et al.*, 2014). Excess N can easily leach out from root zone by heavy rainfall and surplus irrigation which can potentially lead to N deficiency and also contamination of surface and ground water.

Table 2.18: Average N recovery from <sup>15</sup>N-labeled fertilizer application by maize, rice and wheat (Goulding *et al.*, 2008)

Region	Average amount of N fertilizer applied (kg ha <sup>-1</sup> )	N recovery in crop (%)		
		Maximum	Minimum	Mean
Africa	121	59	10	26
Australia	132	77	7	37
Eurasia	117	54	7	31
Europe	156	87	6	43
North America	115	87	6	36
South America	162	86	24	52
South Asia	116	93	7	41

Nitrogen found in the soil in chemical forms with widely different characteristics in terms of availability to plants and susceptibility to losses (Jarvis *et al.*, 2011). Figure 2.15 shows the typical N cycle in agro-ecosystem. Most of the soil nitrogen are bound onto organic matter and will slowly release organic nitrogen and inorganic nitrogen through decomposition process by microorganism for crop use (Dungait *et al.*, 2012; Lees & Quastel, 1946). Therefore, the availability of N for crop is governed by soil microbes.  $\text{NH}_4$  and  $\text{NO}_3^-$  are two forms of N compound that are readily available for crop growth.

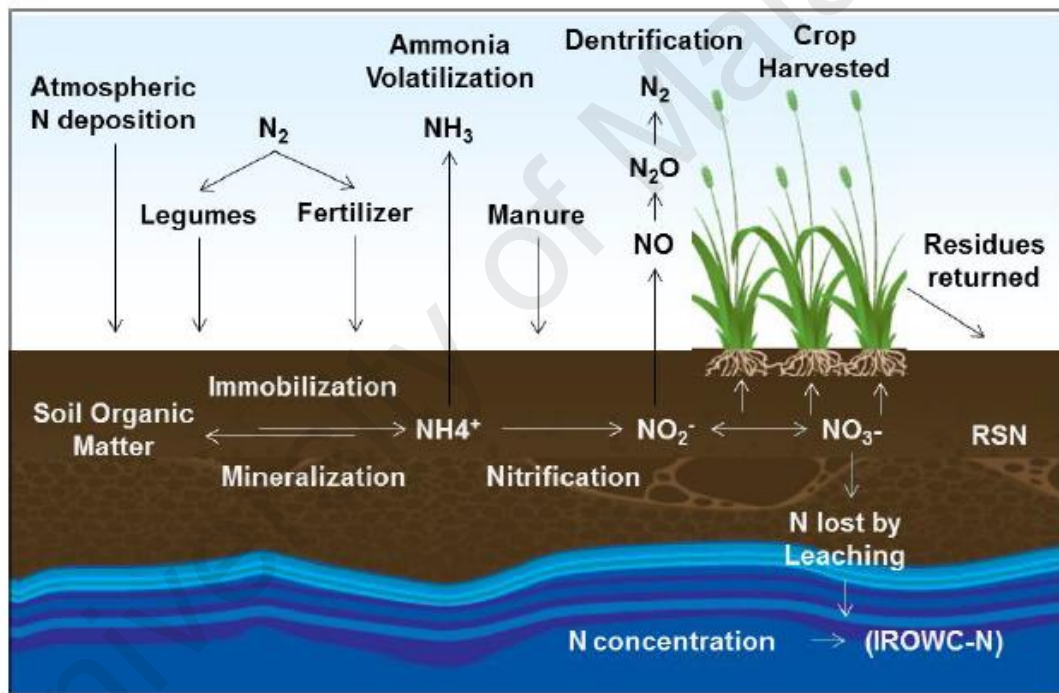


Figure 2.15: Typical N cycle in agro-ecosystem (Yang *et al.*, 2014a)

$\text{NH}_3$  is usually found in the top soil layer which is easily volatile while  $\text{NH}_4$  is usually bound to soil particle (high tendency towards clay mineral) especially under alkaline condition (Jarvis *et al.*, 2011). Nitrate is highly soluble in water which makes it susceptible to loss via runoff and leaching which can lead to pollution of shallow groundwater table (Cao *et al.*, 2014; Wang *et al.*, 2014). Nitrification process converts

ammonium into nitrate form that can be absorbed by plant (Mariotti *et al.*, 1981; Yang *et al.*, 2014a). Nitrogen gaseous emission were often due to nitrification and denitrification processes in the soil. The N management is important in terms of production and also environment as the available N for crop can easily loss to the environment (Jarvis *et al.*, 2011; Rosen & Eliason, 1996).

The soil nitrogen balance has been suggested by OECD and EU as a potential indicator for N leaching and also to monitor farming's environmental performance (DEFRA, 2014b; EPA, 2011; Yli-Viikari *et al.*, 2007). It is also used to monitor the environmental performance after commencement of sustainability program or policy (Brouwer, 1998). Early 19<sup>th</sup> century study shows that the global nutrient balance was recorded in balance or only small surplus (Table 2.19). Unfortunately, the global N surplus increased from 36 Tg N y<sup>-1</sup> in year 1900 to 138 Tg N y<sup>-1</sup> in year 2000 (Bouwman *et al.*, 2013). The increased global mobilization of N due to anthropogenic activities has increased the chances of N loss through NO<sub>3</sub>, NH<sub>3</sub> and N<sub>2</sub>O emission to the environment (Kim *et al.*, 2014). A few studies have examined N balances in vegetable productions and have estimated a 9% to 90% of N surplus (Table 2.20)

Nitrogen balances are affected by several factors in farm management, including farming practice, climate, soil quality, national policies and market condition (Brouwer, 1998). The balance value indicates the N pool that are immobilized by microorganism in the soil or N that are subjected to loss via leaching, volatization, or denitrification to NH<sub>4</sub> or NO<sub>3</sub> (Dungait *et al.*, 2012). However, the N leaching is highly dependent on the climatic conditions and soil properties which complicate the interpretation of the different balance value between various regions (Yli-Viikari *et al.*, 2007). In addition, research also showed the lack of correlation between the N input and N loss by leaching

(Hřebíček *et al.*, 2013). Thus, the N balance value only indicates potential leaching risk but not to estimate the annual N leaching amount (Brouwer, 1998; DEFRA, 2014d). Long term monitoring of farm N balance provides good indication of N balance and leaching (Langeveld *et al.*, 2007). N balance requires a reference value based on research, regulations or targets for the evaluation of system state (Hřebíček *et al.*, 2013). Proper budgeting of soil N input-output can increase the efficiencies of nutrient use within the farm system by better understanding of interaction between soil, plant and water (Dungait *et al.*, 2012).

Table 2.19: Global agricultural N balance timeline

Year		N Source/Input	N Balance
1900–1950	Increase in agricultural production was achieved without synthetic N fertilizers	<ul style="list-style-type: none"> <li>- Fallow periods</li> <li>- Legumes (N<sub>2</sub> fixing crops)</li> <li>- Crop rotations</li> <li>- Recycling of animal manure</li> <li>- Human excreta (Asia countries, eg. China, Korea, and Japan)</li> <li>- Household waste</li> </ul>	N input-output approximately in balance or N surpluses were small
1909	Haber–Bosch process was discovered		
1913	Fertilizer production on an industrial scale Fertilizer use slowly increased in North America and Europe		
1950–2000	The stocks of cattle increased rapidly, particularly in developing countries, production system	<ul style="list-style-type: none"> <li>- Increase fertilizers,</li> <li>- Increase biological N<sub>2</sub> fixation,</li> <li>- Increase animal manure</li> <li>- Increase atmospheric N deposition (NO emissions from industrial activities and fossil-fuel combustion)</li> <li>- Decrease nutrient recovery in crop production system</li> </ul>	Rapid increase in the N surplus
2000–2050	<ul style="list-style-type: none"> <li>- Increasing population</li> <li>- Increase food demand</li> <li>- Human diets shift toward meat and milk</li> </ul>	<ul style="list-style-type: none"> <li>- N input similar with 1950-2000</li> <li>- Nutrient recovery increases rapidly</li> </ul>	28% increase of N surplus

Source: Bouwman *et al.* (2013)



Table 2.20: N input and N balance of various vegetable farms

Soil Types	Country	Crop	N balance (kg N ha <sup>-1</sup> year <sup>-1</sup> )	N input (kg N ha <sup>-1</sup> year <sup>-1</sup> )	References
<b>Aquic Fragiudeot, Typic Epiaguept . Endostagnic Cambisol, Haplic Stagnosol, Gleyed Melanic Brunisol, Orthic Humic Gleysol</b>	Norway	Whear, oat, barley, potatoes	Organic farm: +5.9 to 44.7  Conventional farm: + 16.1 to 63.6	Synthetic fertilizer: 55-141 Manure: 0-110 Rainfall: 7.2 Dry atmospheric deposition: 2	Korsaeth 2008
<b>NA<sup>a</sup></b>	Niger	Vegetables	+1133	Total N: 711 (range: 1109 – 3816)	Diogo et al. 2010
<b>Alluvial</b>	China	Greenhouse vegetables	+3327 kg N ha <sup>-1</sup> year <sup>-1</sup> (range: +620 to +8084)	Total N: 951-8421 Synthetic fertilizer: 1358 Manure: 1881 Irrigation: 402 (range: 4-905)	Ju et al. 2006
<b>NA</b>	China	Greenhouse vegetables	+ 4328	Synthetic fertilizer: 2823 (range: 375- 7344) Manure: 1847 (range: 23-4775 )	Ju et al. 2007
<b>Eutric Fluvisol</b>	Vietnam	Vegetables	+85 to +882	Synthetic fertilizer: 245-808 Manure: 76-91 Irrigation: 7-995 Rainfall: 14-17	Khai et al. 2007

NA: Not available

### 2.13.1 Farm gate N surplus

Tracing the path of N through environmental reservoirs is a considerably challenging because of the complex N cycle, complicated oxidation stages and mechanism conversion, variety of interspecies, and intricate transport/storage processes (Galloway *et al.*, 2004). Thus, some parties prefer the use of farm gate nitrogen surplus which is simpler and convenient to use. The nitrogen surplus is one of the most commonly accepted agri-environment indicator in N management (Glasse *et al.*, 2014; Langeveld *et al.*, 2007). Similar with nutrient balance, N surplus considered the N input-output of the farm system however; the farm output only measures the crop harvested from the system (Eq. 2.1)(van Eerdt & Fong, 1998). The excess N after deducting N exported from the system are assumed to be subjected to potential loss via volatilization, runoff and leaching.

$$\text{N surplus} = \text{Total N input} - \text{total N output via crop} \quad \text{Eq. 2.1}$$

Farm gate N surplus operates as a simple accounting based on readily available data at the farm scale that are likely to be fairly accurate, easy to interpret and practical assessment tool for farmer's daily decision making of routine practices (Gourley *et al.*, 2007). Higher N surplus is considered less efficient than low N surpluses. However, the concept of nitrogen surplus does not reflect variations in complex interrelation between the biological process in soil and the nitrate leaching (Langeveld *et al.*, 2007).

### 2.14 Carbon sequestration

Between 1990 to 2007, the annual carbon sequestration by forest is estimated to be two billion metric tons of carbon ( $2.4 \pm 0.4 \text{ Pg C year}^{-1}$ ) (Pan *et al.*, 2011). Deforestation due to increased population and food demand is one of the main reasons for carbon sink

reduction that results in 20% to 50% loss of this stored C (Eswaran *et al.*, 1993). Commercial agriculture is one of the key driver of deforestation and it is estimated that 50% of the global deforestation in the last decade has been for agriculture purposes (Lawson *et al.*, 2014). Biomass burning, crop production and conversion of grasslands to croplands are the primary human activities that increased atmospheric CO<sub>2</sub>. About 25% to 75% of soil organic carbon in agro-ecosystem were lost (Lal, 2011). In addition, the cultivation process is considered a mining process when soil nutrient is rapidly removed through crop harvesting and also lost to environment which increased the carbon emission (Follett, 2001). Severe soil organic carbon (SOC) depletion leads to soil degradation, productivity, impacts on water quality or even contribute to global warming (Lal, 2004a). The soil degradation is a serious problem in developing countries especially South Asia and sub-Saharan Africa (Oldeman, 1994).

C is mostly stored as organic matter instead of timber wood which includes woody debris, soil, wood products preserved in landfills, and woody plants. About 75% of carbon in US is found in organic matter (Pacala *et al.*, 2001; Wofsy, 2001). The annual carbon sequestration between 1990 and 2007 by forest is equivalent to 25% carbon emission from fossil fuel combustion (Wofsy, 2001). This is estimated to offset 0.4-1.2 Giga tonnes of C emission from fossil fuel combustion (Lal, 2004a). Carbon sequestration is defined as the increase of SOC by appropriate land management with the aim to mitigate climate change (Powlson *et al.*, 2012; Powlson *et al.*, 2011). Carbon sequestration is achieved by high biomass input to increase SOC and soil organic matter (SOM) coupled with soil management that conserve soil structure, reduce disturbance, conserve water, promote nutrient cycling, and enhance microbial and soil fauna diversity (Lal, 2004a). The SOC is the carbon derived from organic sources and SOM is

the mixture of organic material, humus, charcoal and microbial biomass which generally contained 58% of SOC (Stockmann *et al.*, 2013).

The global terrestrial ecosystem contains approximately 2344 Gt of soil organic carbon and it is an ideal reservoir for carbon sequestration and could offset the CO<sub>2</sub> emission of anthropogenic activities (Luo *et al.*, 2010; Piao *et al.*, 2009; Stockmann *et al.*, 2013). The Intergovernmental Panel on Climate Change (IPCC) has identified biomass application as the promising tool to capture and store carbon at terrestrial reservoir (Albrecht & Kandji, 2003; Schlesinger, 1999; Sims *et al.*, 2007; Smith *et al.*, 1997). Carbon sequestration in terrestrial system is important in reducing GHG emission and at the same time act as a sink to store carbon (Schlesinger, 2000). In addition, it is a relatively low cost mitigation of GHG emissions (Alexander *et al.*, 2015).

Agriculture land has the potential to be carbon sink with proper land management by encouraging carbon sequester practices (Figure 2.16) (Freibauer *et al.*, 2004; Luo *et al.*, 2010; Minasny *et al.*, 2012; Ogle *et al.*, 2005; Vleeshouwers & Verhagen, 2002). Crops captured CO<sub>2</sub> from atmosphere via photosynthesis process and the plant C is retained in the soil through plant litter, root material and exudates (Follett, 2001). Farming practices and management exert high influences over soil carbon content and appropriate practices potentially increase soil organic carbon accumulation rate and sequester C from atmosphere (Freibauer *et al.*, 2004; Srinivasarao *et al.*, 2015; West & Post, 2002). A modelling in Europe showed the incorporation of reduced tillage, use of straw, ley cropping, cover crops and conversion of arable land to grassland has the potential to capture 101–336 tonnes CO<sub>2</sub> eq and 549-2141 tonnes CO<sub>2</sub> eq. via SOC sequestration by 2020 and 2100, respectively (Lugato *et al.*, 2014).

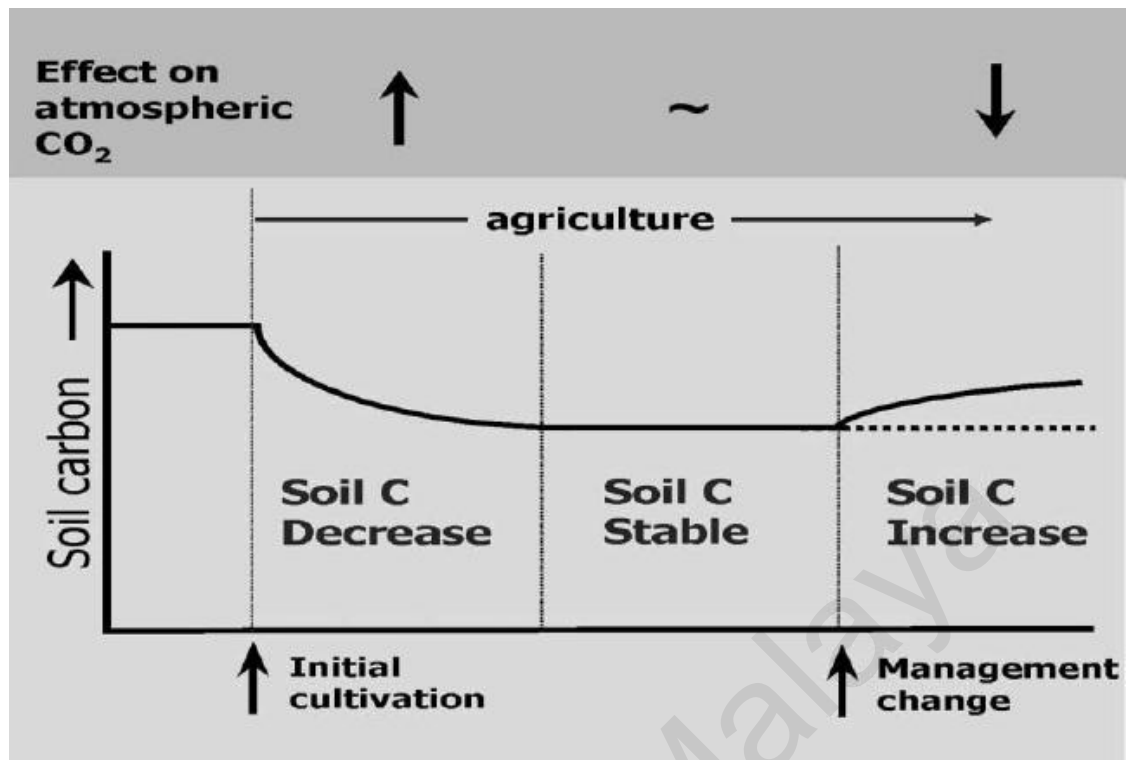


Figure 2.16: Soil C stock change and release of carbon dioxide in agricultural practices (Follett, 2001).

The rate of soil organic carbon sequestration relies on factors such as soil conditions, rainfall, climate, farming system and soil management (Lal, 2004a). Biological processes such as priming effects, biodiversity, root mass and exudates, chemical and physical processes exert high influences over decomposition of SOC (Stockmann *et al.*, 2013). The keys to increase soil carbon sequestration are to increase soil aggregation and stability, macro-porosity, decrease water infiltration rate, reduce evaporation, increase water availability, reduce crusting and compaction, reduce erosion risk, prevent water pollution, increase soil cation-anion exchange capacity, increase plant nutrient reserves, increase microbial carbon, increase methane oxidation capacity, modest nitrification and denitrification rate, reduce leaching, increase soil buffering, capacity, moderate elemental balance, improve production and resource use efficiency (Lal, 2011). Formation of soil macro aggregate is crucial as it enhance the soil C accumulation (Kahlon *et al.*, 2013; Sánchez-de León *et al.*, 2014; Six *et al.*, 2000; Six *et*

*al.*, 2006). Fertilization application of organic matter to increase SOC is one of the key pathway for sequestration of CO<sub>2</sub> in agriculture (Yang *et al.*, 2015). Carbon sequestration from organic matter and crop residue depends on decomposition rate by microbial. Research indicates that only 70% of crop residues on the field are utilized by microbes yearly (Tian *et al.*, 2015). There are several farm practices that has the potential to increase SOC (Follett, 2001):

- Judicious nitrogen input. There is a complex interaction between C and N in soil (Figure 2.17) and research suggested that the decomposition of soil nitrogen will increase soil carbon sequestration (Schlesinger & Andrews, 2000). Research also shows the input of N fertilizers has increased 37% soil C storage after abandonment of the agriculture land (Vuichard *et al.*, 2008),
- Conservation tillage improved soil porosity and water retention capacity which enhance the edaphic environment and efficiency of inputs use (Kahlon *et al.*, 2013). A data analysis of 67 long-term agriculture experiments indicates the conversion from conventional tillage to no-till system can sequester  $57 \pm 14 \text{ g C m}^{-2}$  yearly with a 5-10 years to reach equilibrium state (West & Post, 2002). The increase of SOC with conservation tillage highly depends on yield, it is generally agreed that no carbon sequestration benefit in situation when practicing conservation tillage causes yield reduction (Ogle *et al.*, 2012),
- Soil microbes play an important role in protects soil organic matter losses by forming soil aggregation and influencing C cycling. Practices that increase total microbial biomass, fungal-dominated community structure, thereby enhancing the accumulation of microbial derived organic matter, i.e. crop rotations, conservation tillage, and cover crops (Six *et al.*, 2006),

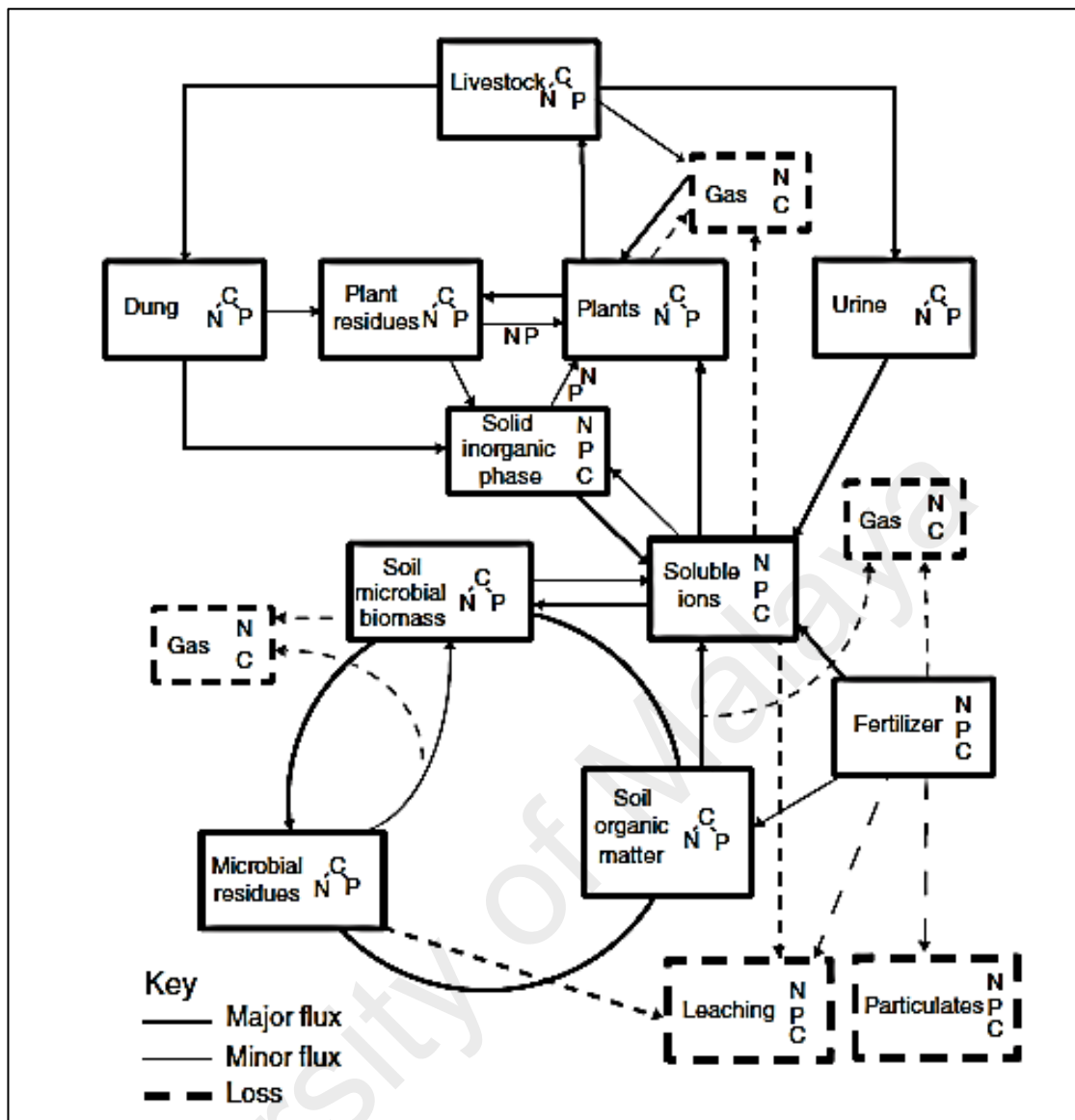


Figure 2.17: Model of the complex interactions of N, P and C cycles in a farming system (Dungait *et al.*, 2012)

- Integrated nutrient management. Appropriate used of livestock manure, straw and compost, coupled with crop rotations that return large quantities of biomass back into farm systems (Jiang *et al.*, 2014; Lal, 2002; Paustian *et al.*, 1992). Study indicates the SOC and crop yield under the combination used of inorganic fertilizer with organic residues are higher compared to the plot that uses only inorganic fertilizer or organic residues (Yang *et al.*, 2015),
- Application of biochar (Mao *et al.*, 2012; Spokas *et al.*, 2012),

- Earthworms contribute to the formation of soil aggregates and is an important factor contributing to the soil stabilization (Sánchez-de León *et al.*, 2014),
- Phosphorus oxide, silicon, titanium and clay particles were associated with potential for promoting C storage (Song *et al.*, 2014; Yunusa *et al.*, 2015),
- Organic residues. The net primary productivity determines the amount of organic residues that returned or remained in the soil after harvest (Leifeld *et al.*, 2013),
- Soil amendment with high stable C and low C:N ratio able to alleviate microbial stress in agricultural soils by transforming from C neutral status to a C sink (Tian *et al.*, 2015),
- Cultivation of cover crop and legumes as green manure (Guan *et al.*, 2016). A meta-analysis of 139 plots at 37 different sites showed cover crop treatments significantly increases SOC stock than reference croplands (Kahlon *et al.*, 2013; Poeplau & Don, 2015), and
- Agroforestry. Model signified a total 1.1–2.2 Pg of C could be removed from the atmosphere by global scale of agroforestry system within 50 years implementation (Albrecht & Kandji, 2003).

Inappropriate farm management can lead to carbon loss from agriculture soil and these include (Corsi *et al.*, 2012; Govaerts *et al.*, 2009; Lal, 2002):

- Soil disturbance
- Mono-cropping
- Poor management of crop residues
- Soil erosion
- Cultivation of upland soils
- Negative nutrient balance in cropland



- Residue removal
- Soil degradation by accelerated soil erosion and salinization
- Soil mining (low biomass input and high biomass output)
- Soil liming increases carbon losses via CO<sub>2</sub> emission (Nadeem *et al.*, 2015)
- Fallow under bare soil condition (Follett, 2001)
- Excessive fertilizers and pesticides use

Carbon sequestration in tropical and sub-tropical regions faced difficulties because of the high soil degradation rate (Lal, 2004b). The soil depletion level is much higher at tropics region indicates high C sink capacity potential and also low sequestration rate (Lal, 2004a). The restoration of degraded soil and ecosystems in tropics and subtropics is much needed. Extensive research on C sequestration has been done but mostly in temperate region (Foerid & Høgh-Jensen, 2004; Ogle *et al.*, 2005). The insufficient information of carbon storage in agriculture land was noticed especially in developing world, tropics and subtropics regions (Govaerts *et al.*, 2009). Data limitation is the main set back in meta-analysis of global soil carbon change (Leifeld & Fuhrer, 2010).

Organic biomass input appears to benefit the soil health, thus organic farming is perceived to preserve and improve soil quality due to the large amount of organic input (Lynch *et al.*, 2012). Conventional farm lack of organic input is viewed to be the contributor to GHG emission while the high organic input in organic farms often seen as C sequester. However, Leifeld *et al.* (2013) challenged the benefit of organic farm in climate change through soil carbon sequestration because organic matter is also widely used in conventional farm and the low yield of organic system may lead to land expansion. Various reports tried to conclude the benefits of converting from conventional farm to organic farm in regards to carbon sequestration (Eve *et al.*, 2002;

Liu *et al.*, 2013b). Data analysis of 74 studies indicates higher C stocks under organic farm system as compared to the conventional farm system (Gattinger *et al.*, 2012). Evidence of higher soil carbon concentration in organically managed farm was found, yet some other studies have not agreed with such findings (Janzen, 2006; Leifeld & Fuhrer, 2010; Scialabba & Müller-Lindenlauf, 2010). Several modelling studies revealed that conversion of CF to OF which increases soil carbon is only a temporary solution for carbon sequestration (Foereid & Høgh-Jensen, 2004; Smith *et al.*, 2001). The inconsistent findings of whether organic farm contributed to carbon sequestration has become an issue of controversy.

There are several limitations of carbon sequestration as climate change mitigation which include the following constraints (Powlson *et al.*, 2011):

- i. Quantity of carbon stored in soil is finite and the carbon sequestration process is reversible,
- ii. The increased of SOC may induce changes in the fluxes of other greenhouse gases, especially nitrous oxide and methane. The efforts to increase carbon sequestration appear to enhance soil's capacity to oxidize CH<sub>4</sub> but also increases the emissions of N<sub>2</sub>O (Nadeem *et al.*, 2015),
- iii. The increase of SOC from conservation tillage, addition of manure, organic residues is low compared to effort such as afforestation and halting deforestation and reducing fossil fuel usage (Smith *et al.*, 1997),
- iv. There are hidden carbon cost in carbon sequestration, e.g the production of fertilizer and fuel used to pump water for irrigation contribute to carbon emission (Schlesinger, 1999; Schlesinger, 2000),
- v. The recommended method like conservation tillage, residue retention, fertiliser N application method increased soil carbon in top soil (10 cm) which is

vulnerable to environmental and management pressures (Lam *et al.*, 2013).

Sampling of deeper soil shows no carbon sequestration advantage in conservation tillage (Baker *et al.*, 2007),

- vi. Raised of temperature due to global warming increases soil C oxidation may result in further increase of atmospheric CO<sub>2</sub> (Stockmann *et al.*, 2013), and
- vii. The efficiency of recommended practices to increase SOC are highly variable due to differences in climate, soil conditions and farm management (Lal, 2011).

The SOC sink capacity and the permanence depends on the soil properties (clay content, mineralogy, stability, microaggregates), landscape and climate (Lal, 2004a). A study indicates proper enhancement in rotation complexity will prolong the time frame before reaching the soil equilibrium stage from 5-10 years to 40-60 years (West & Post, 2002). Thus, improved management of cultivated land could contribute significantly to CO<sub>2</sub> mitigation. Moreover, increasing SOC stocks have additional benefits with respect to enhanced soil fertility, soil water retention, improve soil structure, decrease risks of erosion and degradation, restoration of degraded soil, sustain soil microorganisms and increase agricultural productivity (Lal, 2004a; Schlesinger, 1999; Wiesmeier *et al.*, 2014). The SOC also acts as a bio-membrane that filters and degrades contaminants, reduce loss of sediment to surface water and hypoxia risk in aquatic system (Lal, 2004a). The research by Lal (2004) shows increment of one tonne of SOC increases wheat yield by 20-40 kg ha<sup>-1</sup>, maize yield by 10-20 kg ha<sup>-1</sup>, and cowpeas by 0.5-1 kg ha<sup>-1</sup> (Lal, 2004a). A 22 years experiment in China also demonstrated that the incorporation of inorganic fertilizer and crop residues can efficiently increase crop yield and SOC (Yang *et al.*, 2015). The research also shows a high correlation between crop yield and soil organic carbon mineralization (P<0.05). It is estimated that the improvement increased one tonne C ha<sup>-1</sup> yr<sup>-1</sup> of SOC pool in the root zone can increase food production in

developing countries by 24 to 32 million tonnes of food grains and 6–10 million tonnes of roots and tubers, annually (Lal, 2011). Despite all the limitation, increasing the SOC pool is essential to enhance global food security and environment sustainability development (Lal, 2011). If an agriculture system couldn't attain carbon sequestration then it is often in a stage of losing carbon due to improper management, thus proper farm management is crucial to prevent further loss of C from agriculture soil (Corsi *et al.*, 2012; Govaerts *et al.*, 2009). The understanding of the carbon pool and fluxed in the farm system can provide information to farmers to reduce carbon loss or if not achieving carbon sequestration (Lal, 2009).

#### **2.14.1 C balance and C flux**

Carbon balance and organic matter balance are one of the basic and the most frequently used bio-physical indicators (Hřebíček *et al.*, 2013). Carbon flux and carbon budget is used in The Kyoto Protocol as verification tool on carbon stock change in a system (Steffen *et al.*, 1998). The United Nations Framework Convention on Climate Change (UNFCCC) recognizes the importance of accounting for carbon flux which is defined as the difference between C sequestered in the soil and the total C emissions from all farm inputs and operations (West & Marland, 2002). It is seen as an appropriate tool for long term monitoring of carbon storage in terrestrial system as it reflects the carbon stock changes over time. Carbon balance approaches can also be used to estimate one or more pools or fluxes (Stockmann *et al.*, 2013). Carbon balance uses budgeting method to calculate the carbon input-output of a system to determine whether it is a carbon sink or source (Hongyeng & Agamuthu, 2014; Lal, 2011). There has been concern on the uncertainties in estimating carbon flux but with appropriate methodology the uncertainties level can be under the acceptable levels (Steffen *et al.*, 1998).

## 2.15 Material/substance flow analysis

Material/substance flow analysis (MFA/SFA) has been widely used to trace the flow of production, use, and consumption of materials or element for various economic sectors and discipline, e.g. industrial management, industrial ecology, waste management, architecture, ecological, energy, environment and agriculture (Bailey *et al.*, 2004; Brunner & Rechberger, 2004; Davis *et al.*, 2007; Fuse & Tsunemi, 2012; Guo *et al.*, 2015; Hashimoto *et al.*, 2007; Hawkins *et al.*, 2007; Huang *et al.*, 2012; Huang *et al.*, 2006b; Huang *et al.*, 2013; Lau *et al.*, 2013; Nakajima *et al.*, 2013; Nakamura & Nakajima, 2005; Sendra *et al.*, 2007; Smit *et al.*, 2015). It describes the input-output model of materials and elements caused by economic activities which have the capability to estimate direct and indirect environmental impact (Wohlgemuth *et al.*, 2006). This is crucial for effective environmental management.

The MFA/SFA is often used as policy decision making tool in resources and environmental management by highlighting imminent resources or environmental problems without depending on environmental stress signals (Lau *et al.*, 2013). It assists in improving the effectiveness and sustainability of resource management via early recognition of potential wastage or pollution by showcasing the linkage between anthroposphere system with the environment through examinations of the loading to material (Hendriks *et al.*, 2000). It has become a useful tool as environment indicator, eco-efficiency indicator and industrial ecology (Sendra *et al.*, 2007; Wang *et al.*, 2016). It offers a broad scope of application in environmental accounting and systems analysis, and it has appeared to be the principal practical framework across various flow accounting methods (Ulhasanah & Goto, 2012). The MFA/SFA also has the potential to act as a guide for regional audit and environment management. In Japan the MFA/SFA

is proposed as the indicator of material inputs and outputs across the national system (Hashimoto & Moriguchi, 2004).

The MFA/SFA is a standardized methodology for accounting the input and output material flows of a system that quantify the material within a system (Huang *et al.*, 2012; Sendra *et al.*, 2007). It can be used to model nutrient balance or farm-gate budget which records the amounts of nutrients from various materials being imported and exported into the farm system (Figure 2.18) (Oenema *et al.*, 2003). Like ecosystem, a farm system also has a unique characteristic and metabolism. The metabolism of a farm system can be managed to achieve sustainable resource management and environmental development (Jakrawatana *et al.*, 2015). The advantages in utilizing MFA/SFA for farm level analysis is its ability to quantify all material flows and identifying significant and simple indicators that can discover critical points and demonstrate the metabolism state and changes of the farm system (Sendra *et al.*, 2007). By analysing the material flow of a farm system, it provides the farmers with better understanding of the functioning of their farm systems (Smit *et al.*, 2015). The results generated from MFA/SFA can be used to evaluate the system based on reference value from previous research, agri-environment indicator or national regulation (Hendriks *et al.*, 2000). In general, the basic frameworks of material flow analysis are:

- Farm system material flow analysis,
- Evaluate the results and identify important and relevant flow and stock, and
- Management of material flow and stock to achieve objectives in sustainable view.

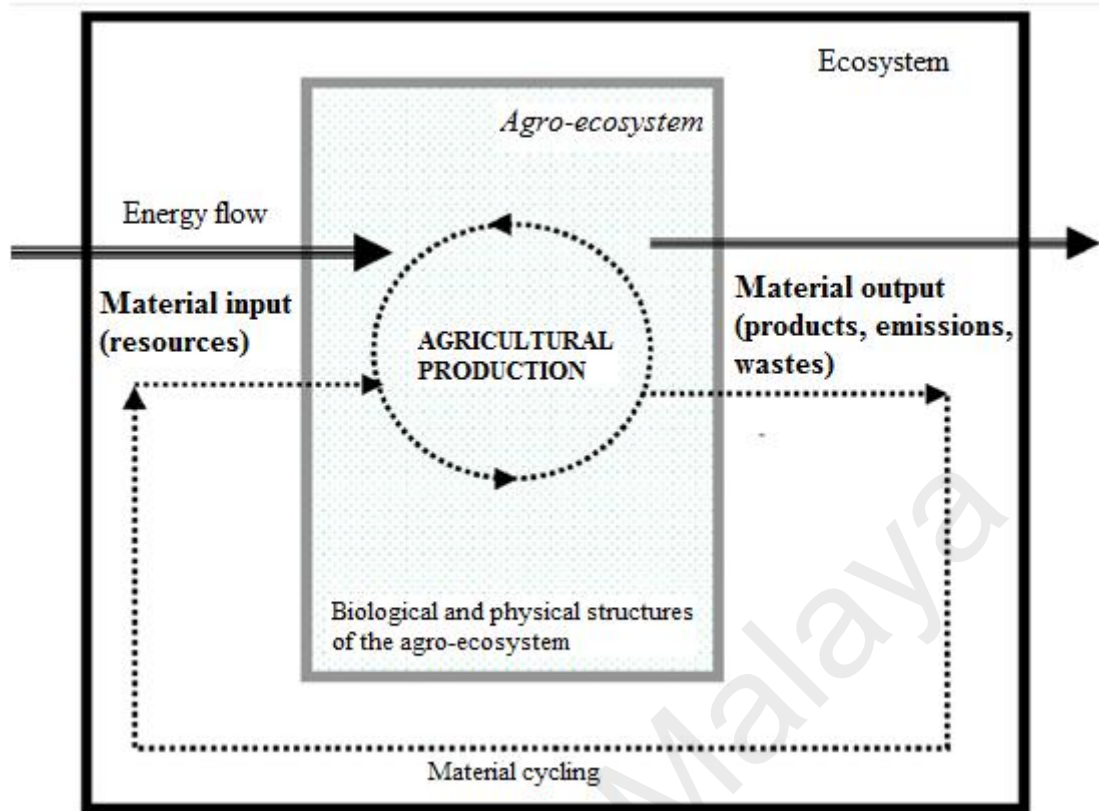


Figure 2.18: Flow indicator and state indicator of agro-ecosystem (Yli-Viikari *et al.*, 2007)

The MFA/SFA is a systematic assessment of mass flow within defined spatial and temporal boundaries (Cencic & Rechberger, 2008; Habib *et al.*, 2014). The basis of MFA/SFA is based on the laws of mass and energy conservation where the input is equal to the sum of output and materials that accumulate within the system (Lau *et al.*, 2013). The system is considered a black box in which only input and output are viewed (Rincón *et al.*, 2013). The dynamic approach of MFA/SFA is the analysis the flows of materials or any stock accumulation over a period of time based on mathematical probabilistic distributions (Chen *et al.*, 2012). The aim of MFA/SFA is to describe and analyse a system as simple as possible, but detailed enough to allow a good understanding of a system for user to have a good control of the system management (Cencic & Rechberger, 2008; Espinoza *et al.*, 2014). The basis of process evaluation is

the entire balance of materials which can be described with mass balance (Eq. 2.2) (Rotter *et al.*, 2004).

$$\text{Mass balance} = \text{Input} - \text{Output} \quad \text{Eq. 2.2}$$

Generally, mass flow is quantified based on six major steps of material flow analysis (Brunner & Rechberger, 2004; Espinoza *et al.*, 2014; Herva *et al.*, 2012; Sendra *et al.*, 2007):

- i. Definition of research objective and selection of substances,
- ii. System definition in space boundaries and time frame,
- iii. Identification of stocks, process and flow within the system boundaries,
- iv. Determination of mass flows and stocks through data collection,
- v. Design material flow chart, and
- vi. Assessment and interpretation of material/substance flow and stocks.

Limitations of MFA are (Cencic & Rechberger, 2008; Huang *et al.*, 2012):

- i. Issues with standardized units and aggregation techniques, and
- ii. Uncertainties or inconsistent data.

### **2.15.1 Sink or source**

The modeling of material flows and stocks provide basis for resource management or environmental pollution control (Laner *et al.*, 2014). The environmental stock is an asset or capital that provides material or economic goods. A stock can be a “source” of inputs or as “sink” for mass (van der Werf & Petit, 2002). The analysis of material flow determines the system balance and stock. The mass change over a period of time is used to classify a process as “source” or “sink” within a farm system (Eq. 3) (Kellner *et al.*,



2011). The term ‘sink’ is defined according to Kral and Brunner (2013), where it is a process with a positive stock change.

$$\Delta, farm = \int_{t_0}^t m, input (\tau) d\tau - \int_{t_0}^t m, output (\tau) d\tau = \begin{cases} > 0 : sink \\ 0 : equilibrium state \\ < 0 : source \end{cases}$$

Eq. 2.3

where,  $\Delta$  is the change in mass over period of time,  $\int_{t_0}^t m, input (\tau) d\tau$  is the input flux rate while  $\int_{t_0}^t m, output (\tau) d\tau$  is the output flux rate.

A positive nitrogen balance indicates potential loss of nutrient to the environment or accumulations in the soil while negative balance signifies soil nutrient depletion (Bouwman *et al.*, 2013; Dungait *et al.*, 2012; HongYeng & Agamuthu, 2015). Carbon has the potential to be stored in terrestrial system thus agriculture soil has the potential to be carbon sink (Beddington *et al.*, 2012b). The positive carbon balance signifies carbon stock in the system while negative carbon balance indicates carbon loss (Hongyeng & Agamuthu, 2014; Lal, 2011).

## 2.16 STAN 2.5 software

The free software STAN 2.5 and made available since 2006 was developed by the Vienna University of Technology. The word STAN is the short form for subSTance flow ANalysis. The software supports the performing of MFA/SFA according to the Austrian standard ÖNORM S 2096 (Material flow analysis - Application in waste management) under consideration of data uncertainties (Cencic & Rechberger, 2008; Laner *et al.*, 2014). The software development was sponsored by the Austrian Ministry

of Agriculture, Forestry, Water and Environment and The voestalpine Group. The software can be downloaded from <http://www.stan2web.net/>.

The STAN 2.5 combines all necessary features of MFA in one software product: graphical modelling, data management, calculations and graphical presentation of the results (Figure 2.19). The program can visualize the material and substance flows and provides an assessment of all inputs-output and stocks of a system (Habib *et al.*, 2014; Smit *et al.*, 2015). Graphical model was constructed based on predefined process, flows, subsystem and coefficient (Cencic & Rechberger, 2008). The STAN 2.5 is based on basic mathematical model of mass balance as below:

Balance equation:

$$\Sigma \text{ inputs} = \Sigma \text{ outputs} + \text{change in stock} \quad \text{Eq. 2.4}$$

Transfer coefficient equation:

$$\text{Output} = \text{transfer coefficient}_{\text{output}} \times \Sigma \text{ inputs} \quad \text{Eq. 2.5}$$

Stock equation:

$$\text{Stock}_{\text{Period } i+1} = \text{stock}_{\text{Period } i} + \text{change in stock}_{\text{Period } i} \quad \text{Eq. 2.6}$$

Concentration equation:

$$\text{Mass substance} = \text{mass of good} \times \text{substance concentration} \quad \text{Eq. 2.7}$$

The software computes the uncertainties through error propagation and statistical tests to identify random errors which reconcile the measured values (Cencic & Rechberger, 2008). Typical terms used in STAN 2.5 are (Cencic & Rechberger, 2008; Hendriks *et al.*, 2000; Rincón *et al.*, 2013):

- **Goods** are defined as entities of thing, material or immaterial with or without economic value which contains substances, e.g. waste, plastic, compost, oil, air, precipitation (OED, 2010b).

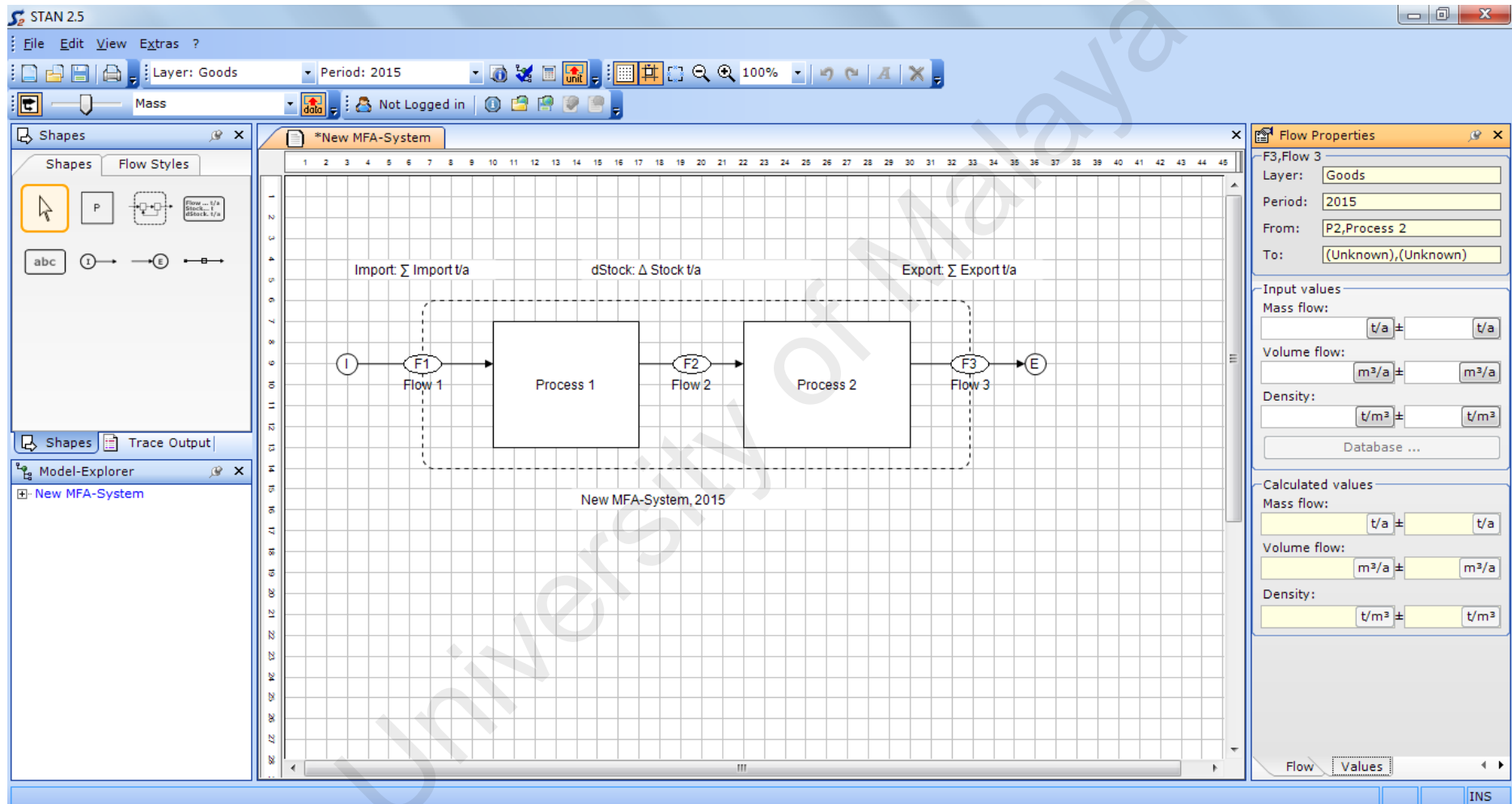


Figure 2.19: User Interface of STAN 2.5 software

- **Substance** is any chemical composition as element or compound, i.e. N, C, Cu, NH<sub>4</sub><sup>+</sup>, CO<sub>2</sub> (OED, 2010e)
- **Process** is placed within a system where a series of operations, activity, transformation, transport or storage activities of materials (OED, 2010c). A “process” is defined as black boxes whereby inputs and outputs are considered and if a process is defined as a subsystem, it indicates that the “process” contains sub-processes.
- **Flow** is defined as material movement from one process to another (OED, 2010a). When the flow crosses the system boundary from outside into the system it is defined as import while the flow crosses the system boundary from inside out of the system it is defined as export flow.
- **Stock** is defined as quantity material or immaterial accumulated via loading of material into a system (OED, 2010d). These stocks can be either potential environmental problems (e.g. hazardous materials) or potential resources (e.g. urban mining).

### 2.17 Farm management efficiency

Better farm management will benefit both the environment and economy. The improvement of farm efficiencies such as nutrient, water and pesticide use are necessary to maintain the productivity and sustainability of farm system while at the same time reduce environmental impact (Tilman, 1999). Integrated efficiency assessment of different farm systems will reveal the impact of farm management practices on the productivity and profitability of farm systems (Cortez-Arriola *et al.*, 2014; Mutoko *et al.*, 2014). Several farm efficiency indicators commonly used to assess the farm management efficiency are:

- Nutrient use efficiency (NUE). The NUE has become a critical part of sustainable agriculture (Goulding *et al.*, 2008). Inappropriate use of nutrient may lead to environmental pollution through the emission of GHG and particulate such as nitrogen and carbon entering water bodies through runoff and leaching (Dungait *et al.*, 2012). By improving the NUE, farmer can reduce the fertilization cost and in the same time reduce or even prevent pollution to the environment. The NUE is considered a potential assessment tool for site specific nutrient management to reduce non-point source nutrient pollution (Buckley & Carney, 2013). This is because the pollution impact is highly farm specific and each farm system is dynamic from one another due to the complex interactions between soil, water, farm practices and atmosphere (Reidsma *et al.*, 2012; Wesström *et al.*, 2014).
- Water use efficiency. Agriculture activities use high volume of water and the global water scarcity has made efficiency water use a priority in farm management. In addition, the excess irrigation or rainfall water wash off agrochemical and nutrient which exit farm system as runoff and leachate is a major source of non-point pollution to surface and underground water (DEFRA, 2015a; Ribaud *et al.*, 1999; UNESCAP, 2014; Zainudin, 2010).
- Yield. Commercial agriculture is the key driver of deforestation, thus higher yield per area of land is assumed to require lesser land to achieve production goal which reduces land expansion and impact on the environment (Alexandratos & Bruinsma, 2012; Lawson *et al.*, 2014). Organic agriculture often has an average of lower yield as compared to conventional agriculture (Laboski & Peters, 2012; Leifeld *et al.*, 2013; Lynch *et al.*, 2012; Razak & Roff, 2007). However, high yields in conventional farming are often involved in exploitation of land which is not sustainable (Bruinsma, 2003).

- Cost profit analysis. Sustainable farm system aims to achieve a balance between the environment, economic and social (Duesterhaus, 1990). Thus, the profitability of farm production system is another concern in sustainable production (Cortez-Arriola *et al.*, 2014). Cost profit analysis assesses the profitability and total cost of conventional and organic farms. There are debates on whether organic farm or conventional farm has higher gross margins because higher profit margin is one of the main factor farmers considered during farm management (Lynch *et al.*, 2012).

University of Malaya

## CHAPTER 3: MATERIALS AND METHODS

### 3.1 Introduction

Farms are dynamic due to variations and complex interrelations of soil properties, climate, nutrient regime, farm management, crop variety, and landscape. To understand a farm system, combination of site survey, field data collection and chemical evaluations are necessary to provide a comprehensive insight of farm metabolism. This chapter explains the methodology and materials used in this study. The primary work of this study includes farm investigation and characterisation, farm input-output survey and chemical analysis, material flow, C flow and N flow modelling, farm efficiency assessment and cost volume profit analysis. This chapter is designed to:

- understand the characteristics and components of organic and conventional vegetable farm systems in Malaysia,
- develop the material, C and N flow model of farm systems,
- identify evidence of potential resource depletion or pollution of farm system,
- identify potential C sequestration in farm systems, and
- identify possible ways to improve farm management towards sustainability.
- evaluate pathogen level in farm systems

### 3.2 Material/substance flow analysis

Material and substance flow analysis (MFA and SFA) is the main methodology framework of this study. The MFA/SFA is a systematic assessment of inputs–outputs within a farm system based on the law of mass conservation where the total farm inputs is equal to the sum of total outputs and stock that accumulates in the system (Eq. 3.1)

(Baccini & Brunner, 1991). Material flow analysis is defined based on law of mass balance and comprises the following fundamental steps described in section 3.2.1, 3.2.2, 3.2.3, 3.2.4 and 3.2.5 (Baccini & Brunner, 1991; Brunner & Rechberger, 2004; Hongyeng & Agamuthu, 2014).

$$\Sigma \text{ inputs} = \Sigma \text{ outputs} + \Sigma \text{ change in stock} \quad \text{Eq.3.1}$$

### **3.2.1 Definition of objective**

Over the years, there has been a debate between organic and conventional farming system on sustainability and contribution to climate change (Bavec & Bavec, 2014; Goulding *et al.*, 2008; Hartz, 2006; Killebrew & Wolff, 2010; Lynch *et al.*, 2012). Therefore, this study is aimed to understand the total input, output and inventory of the studied organic and conventional leafy vegetable farms with material flow analysis.

Annual C and N load data from farm scale studies were included in this research. N was chosen because N is the main factor that controls crop productivity and excessive use of N could lead to serious environmental pollution (Ju *et al.*, 2006; Rosen & Allan, 2007; Rosen & Eliason, 1996). In addition, nitrate and nitrite also impact drinking water quality and are listed as primary drinking water pollutants by USEPA (Harmel *et al.*, 2006). Carbon was chosen due to high loss of C from agriculture soil leads to soil which degradation and affects crop production. Thus, the development of improved agronomic practices is encouraged in order to increase soil C stock. Furthermore, agriculture system has the potential to be C sink by C sequestration which contribute to climate change mitigation (Lal, 2011). C and N flux in farm systems is an important environment indicator. Therefore, this study aims to determine the metabolism of C and



N flux within organic and conventional farm systems and to identify imminent environment issues. The study was conducted from January 2012 to December 2014.

### 3.2.2 Definition of system boundaries and time frame

The system boundaries in this study are within two conventional (CF1 and CF2) and two organic (OF1 and OF2) vegetable farms located in two major lowland vegetable production areas at Selangor state (central region) and Johor state (southern region) in Malaysia (Plates 3.1-3.4).



Plate 3.1: Satellite image CF1 (Source: Google earth, imagery date 23/1/2010)



Plate 3.2: Satellite image CF2 (Source: Google earth, imagery date 14/9/2014)



Plate 3.3: Satellite image OF1 (Source: Google earth, imagery date 30/3/2011)



Plate 3.4: Satellite image OF2 (Source: Google earth, imagery date 16/5/2012)

The climate in the study areas is classified as “equatorial” by the Malaysian Meteorological Department. All four farms have similar climatic conditions with high humidity, and temperature ranging from 25 °C to 30 °C (Figure 3.1-3.4). Average for rainfall during the study period is  $224 \pm 130$ ,  $201 \pm 95$ ,  $235 \pm 155$ ,  $170 \pm 90$  mm mth<sup>-1</sup> for CF1, CF2, OF1, and OF2, respectively (Figure 3.5-3.8). The number of rainy days ranged between 160 and 218 days y<sup>-1</sup> (Table 3.1-3.4). The study farms have distinctive characteristics concerning their farm management practices which is summarized in Tables 3.5 and 3.6.

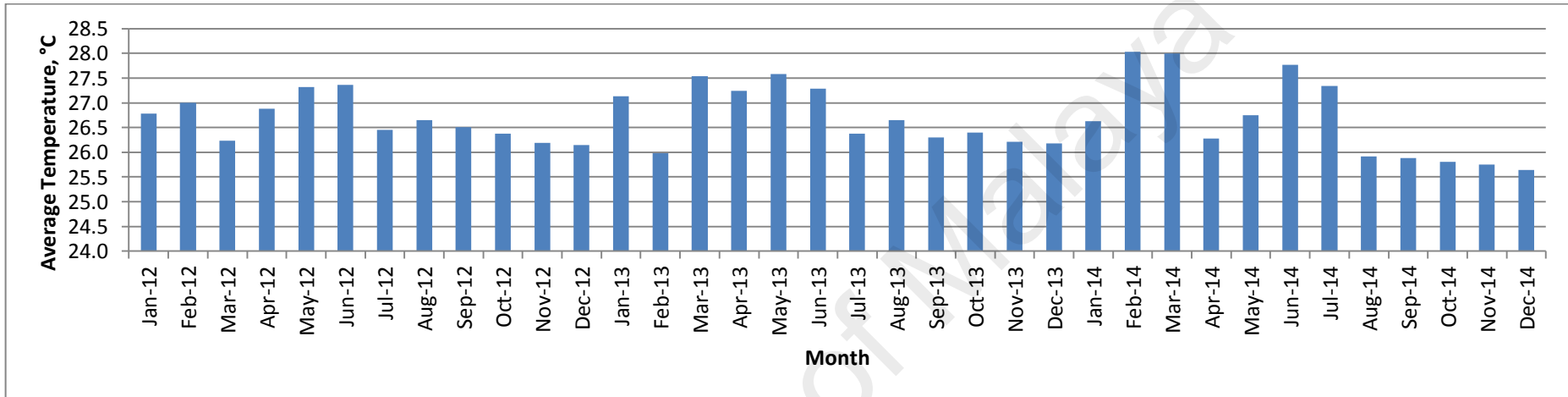


Figure 3.1: Average temperature during study period at CF1

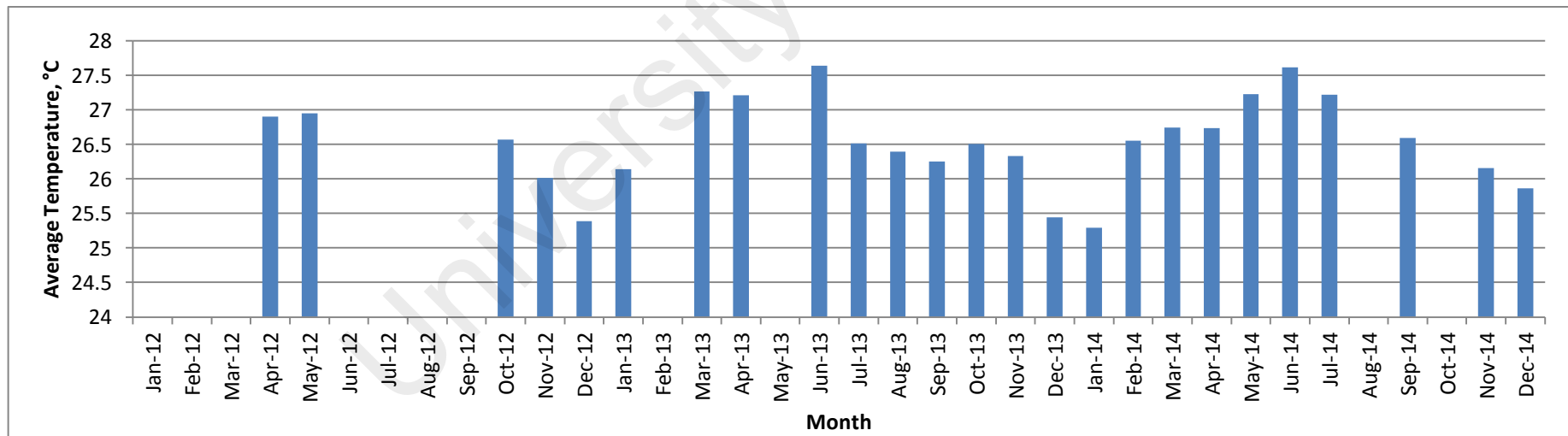


Figure 3.2: Average temperature during study period at CF2

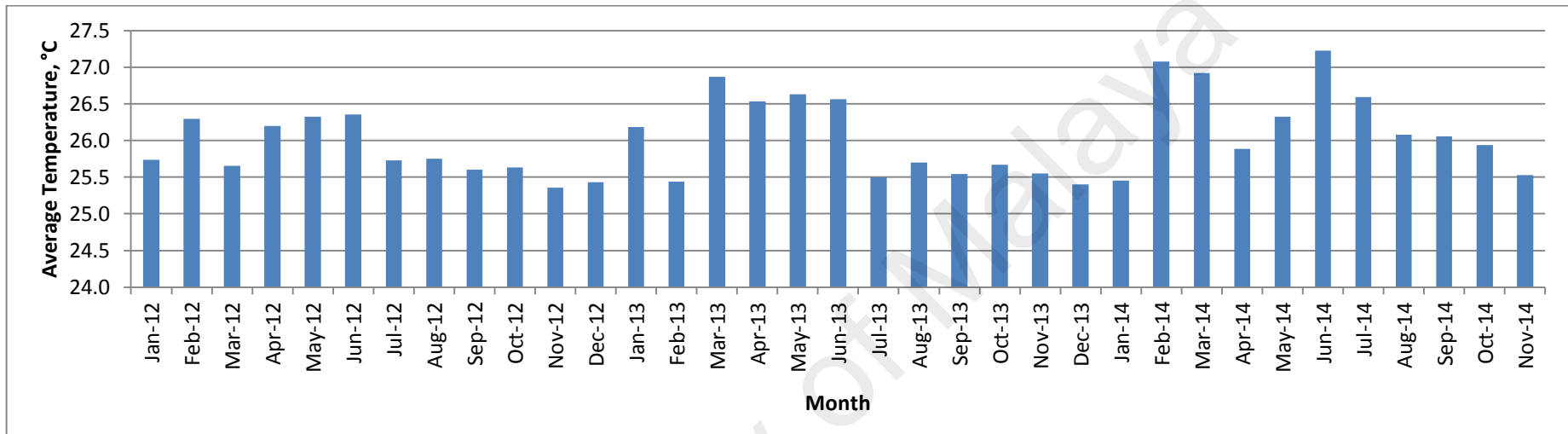


Figure 3.3: Average temperature during study period at OF1

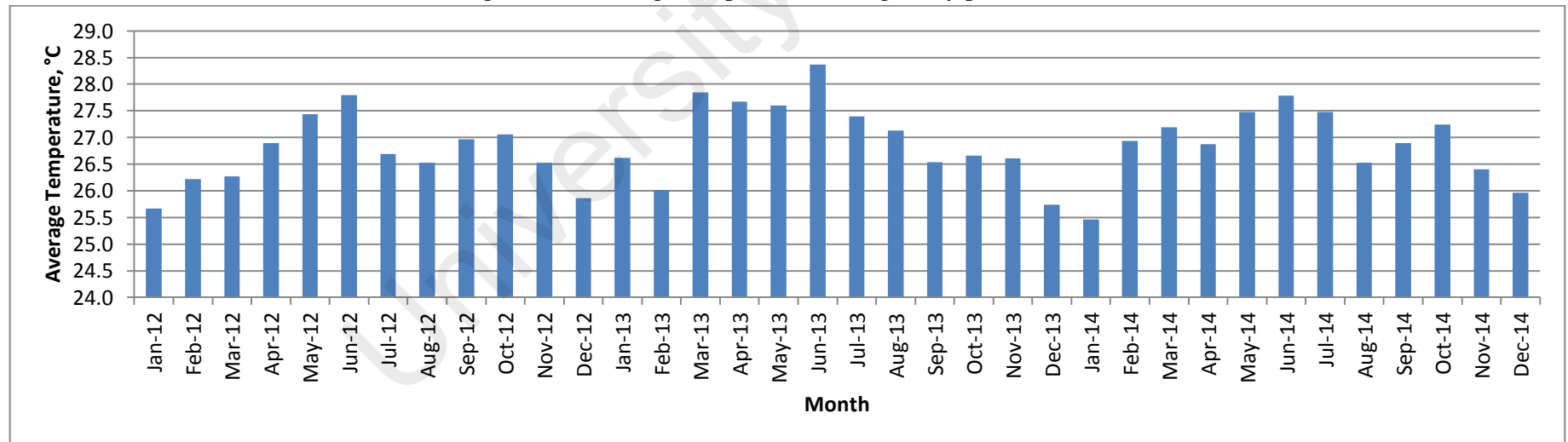


Figure 3.4: Average temperature during study period at OF1

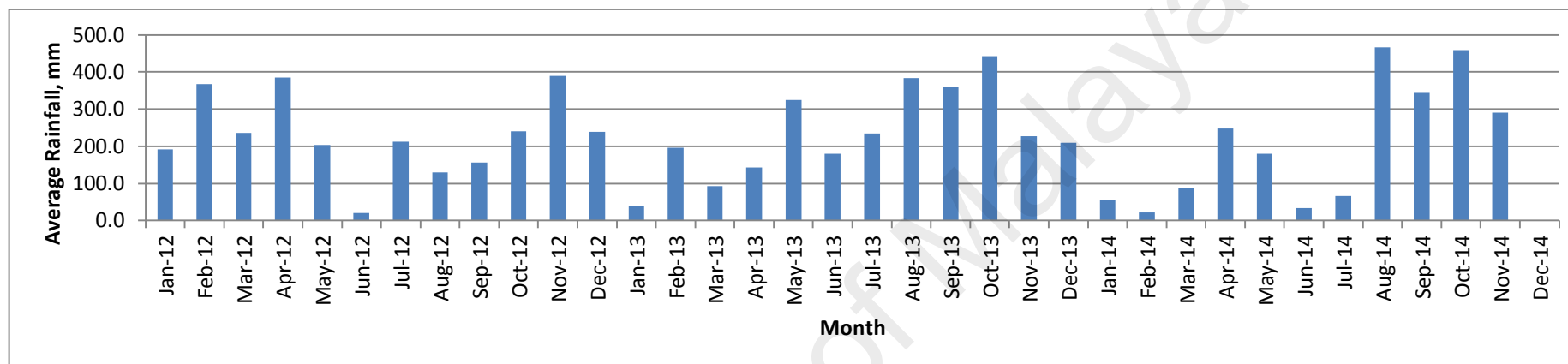


Figure 3.5: Average rainfall during study period at CF1

Table 3.1: Number of rainy days during study period at CF1

Year	Month												ANNUAL
	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	
2012	12	19	18	18	15	4	15	17	12	23	26	21	200
2013	12	20	17	22	22	14	16	21	13	21	22	19	219
2014	6	3	12	24	21	8	10	18	24	26	21	Def.	Def.

\*Meteorological station: Hospital Kuala Kubu Baru (Latitude: 3° 34' N, Longitude: 101° 39' E, Elevation: 61.0 m)

N.A. - Not Available

Def. - Defective Value

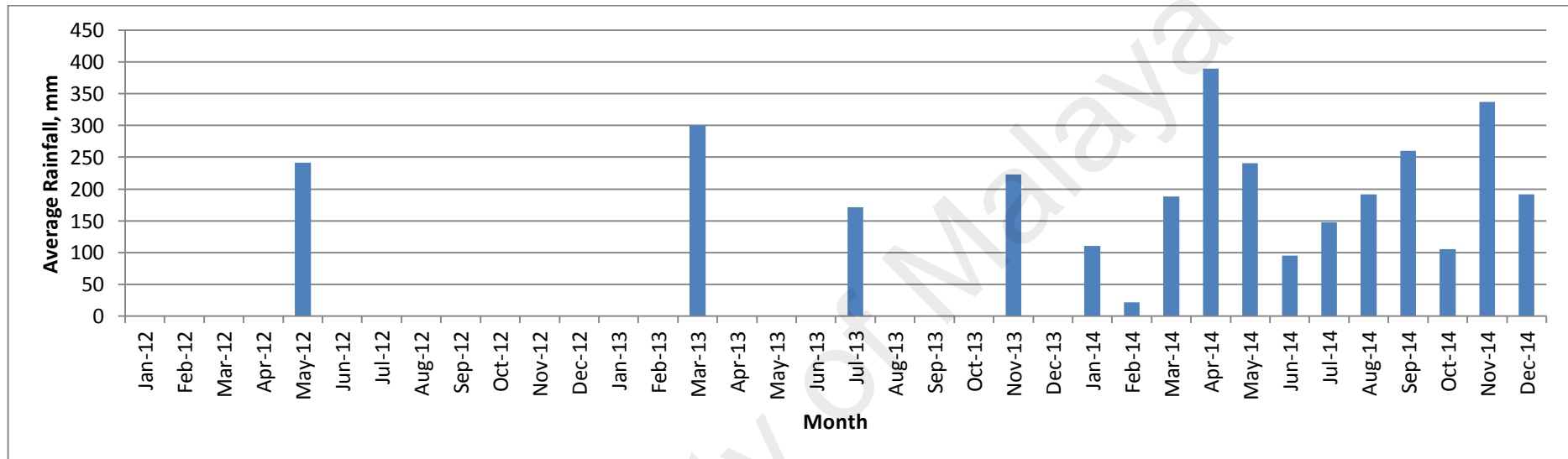


Figure 3.6: Average rainfall during study period at CF2

Table 3.2: Number of rainy days during study period at CF2

Year	Month												ANNUAL
	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	
2012	Def.	Def.	Def.	Def.	15	Def.	Def.	Def.	Def.	Def.	Def.	Def.	Def.
2013	Def.	Def.	18	Def.	Def.	Def.	10	Def.	Def.	Def.	21	Def.	Def.
2014	4	4	14	21	22	12	17	9	12	13	23	18	169

\*Meteorological station: Station: Felda Bukit Batu (Latitude: 1° 42' N Longitude: 103° 26' E Elevation: 27 m)

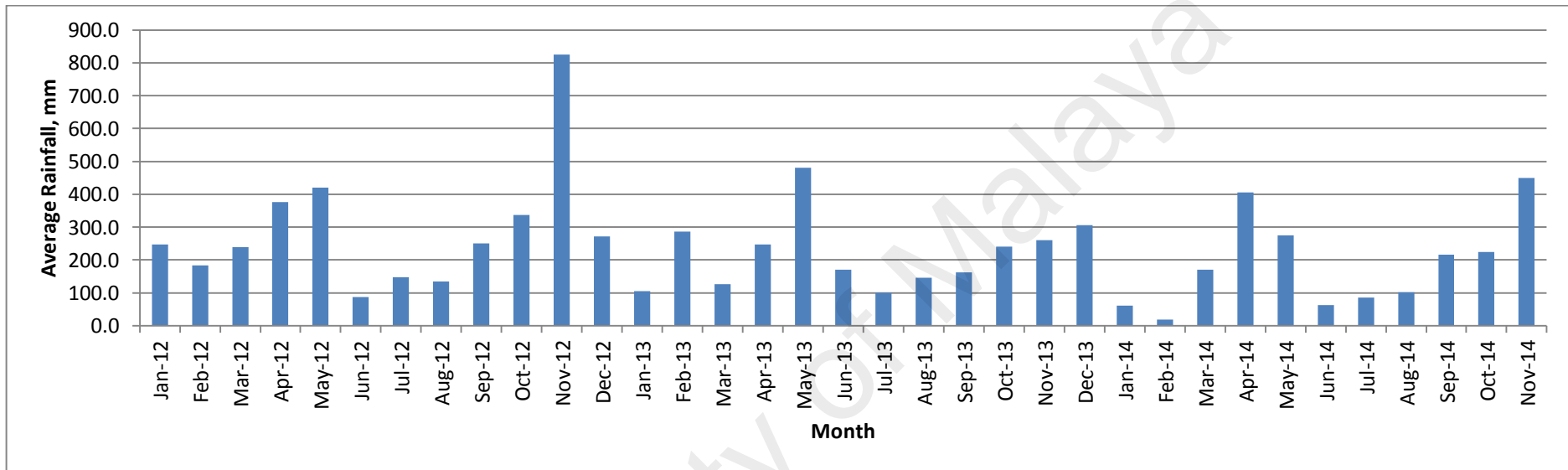


Figure 3.7: Average rainfall during study period at OF1

Table 3.3: Number of rainy days during study period at OF1

Year	Month												ANNUAL
	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	
2012	15	10	19	21	19	14	17	16	16	22	28	21	218
2013	15	19	13	18	18	14	19	21	14	21	23	19	214
2014	24	4	15	28	25	18	23	15	15	19	23		

\*Meteorological station: Ampangan Air Sg Semenyih (Latitude: 3°04'34"N Longitude: 101°52'51"E Elevation: 0 m)

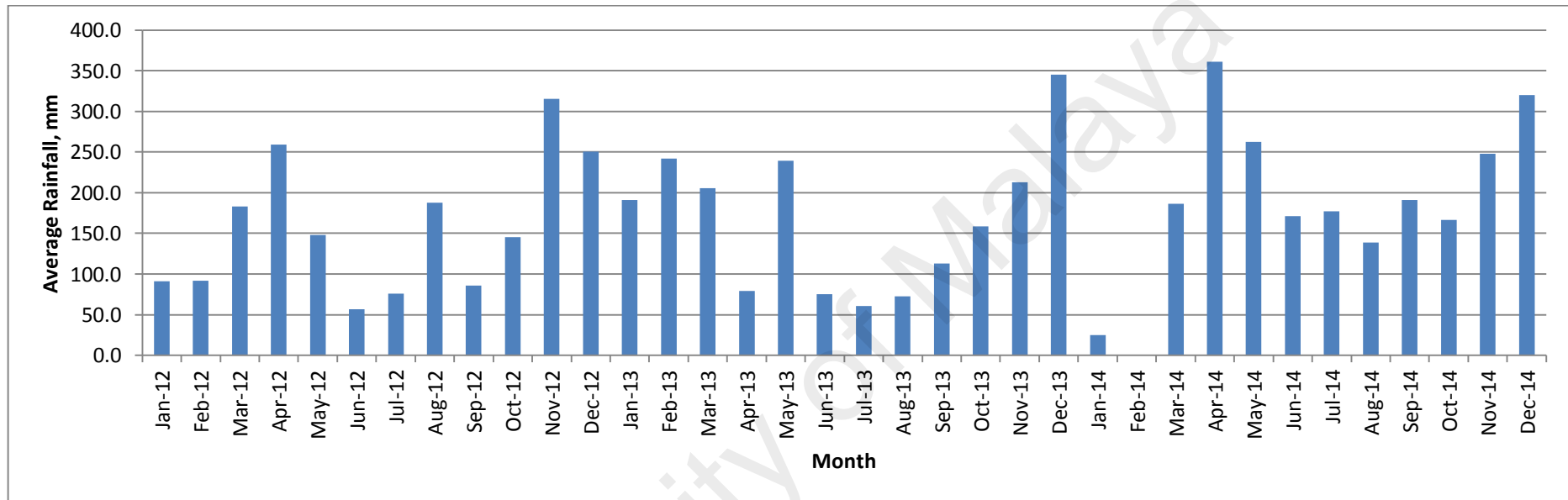


Figure 3.8: Average rainfall during study period at OF2

Table 3.4: Number of rainy days during study period at OF2

Year	Month												ANNUAL
	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	
2012	14	13	18	17	15	7	14	16	9	14	22	24	183
2013	7	17	12	17	13	10	10	19	16	20	16	18	175
2014	3	0	8	21	18	13	13	20	14	14	16	20	160

\*Meteorological station: Kluang (Latitude: 2° 01' N Longitude: 103° 19' E Elevation: 88.1 m)



Table 3.5: Selected farm characteristic of the two organic and the two conventional vegetable farms (between 2012 and 2014)

Characteristics and description	Farm Types <sup>a</sup>			
	CF1	CF2	OF1	OF2
<b>Location</b>	Central region Malaysia: Hulu Selangor (3°25'52.66"N, 101°38'54.50"E)	Southern region Malaysia: Bukit Batu Johor (1°44'59.30"N, 103°24'68.20"E)	Central region Malaysia: Semenyih Selangor (2°56'56.59"N, 101°53'25.69"E)	Southern region Malaysia: Kluang Johor (1°57'76.80"N, 103°12'73.80"E)
<b>Elevation</b>	53 m	33 m	86 m	13 m
<b>Soil Types</b>	Mining reclamation area, (No soil classification available, however, soil composition is mainly clay and sand)	Durian Series (Plinthaquic Paleudult, clayey, mixed, isohyperthermic)	Rengam series (Typic Hapludult, Clayey, Kaolinitic, Isohyperthermic, Haplic Acrisol)	Gong chenak series (Plinthaquic, paleudult, clayey, kaolinitic, isohyperthermic)
<b>Average Temperature (°C)</b>	26.7	26.5	26.0	26.8
<b>Average Rainfall (mm mth<sup>-1</sup>)</b>	224.5	200.9	234.9	170.3
<b>Average number of rainy day (days per year)</b>	210	169	216	173
<b>Leafy vegetable planting area (ha)</b>	5.7	10.1	0.4	1.859
<b>Average Production (kg ha<sup>-1</sup> y<sup>-1</sup>)</b>	107,171	10,693	34,561	114,191
<b>Year of establishment</b>	10 years	2 ½ years	3 years	9 years
<b>Certification</b>	MyGAP	No	SOM	NASAA and SOM

Table 3.6: Selected farm management of the two organic and the two conventional vegetable farms (between 2012 and 2014)

Cultivation practices	Farm Types <sup>a</sup>			
	CF1	CF2	OF1	OF2
<b>Open planting or greenhouse</b>	Open planting	Open planting	Greenhouse	Greenhouse
<b>Irrigation practices</b>	Sprinkler system * Twice a day and each time 5-10 minutes (Dry days only)	Sprinkler system * Twice a day and each time 5-10 minutes (Dry days only)	Sprinkler system * Twice a day and each time 5-15 minutes (Daily)	Sprinkler system * Twice a day and each time 5-10 minutes (Daily)
<b>Soil tillage</b>	Plough with disc or chisel (100mm soil depth) and moldboard (200mm soil depth)	Plough with chisel (100mm soil depth)	Plough with chisel (100mm soil depth)	Plough with disc (100mm soil depth)
<b>Crop rotation</b>	Yes	Yes	Yes	Yes
<b>Soil fallow</b>	Four weeks	Non	Two weeks	Two weeks
<b>Types of fertilizer used</b>	Chemical fertilizer, compost and chicken manure	Chemical fertilizer and chicken manure	Compost and Bokashi compost	Compost
<b>Pest Control</b>	Chemical	Chemical	Pest repellent	Mechanical and pest repellent
<b>Weed Control</b>	Chemical	Chemical	Manual	Manual and thermal

<sup>a</sup> CF1 conventional farm located at central region Malaysia, CF2 conventional farm located at southern region Malaysia, OF1 organic farm located at central region Malaysia, OF2 organic farm located at southern region Malaysia

The farming process varies from farm to farm with different fallow period, fertilization, pesticide, and herbicide application method and timing. Figure 3.9 shows the farming process at CF1. After each harvesting the soil was allowed to fallow and rest for one month before the next crop cycle. One to two weeks before seeding, the soil was tilled and herbicide was applied to remove weed. After seeding, chicken manure was applied immediately. Fertilizer and compost was applied one week after seeding. In general pesticide was sprayed every 5-7 days depending on climate and pest incident.

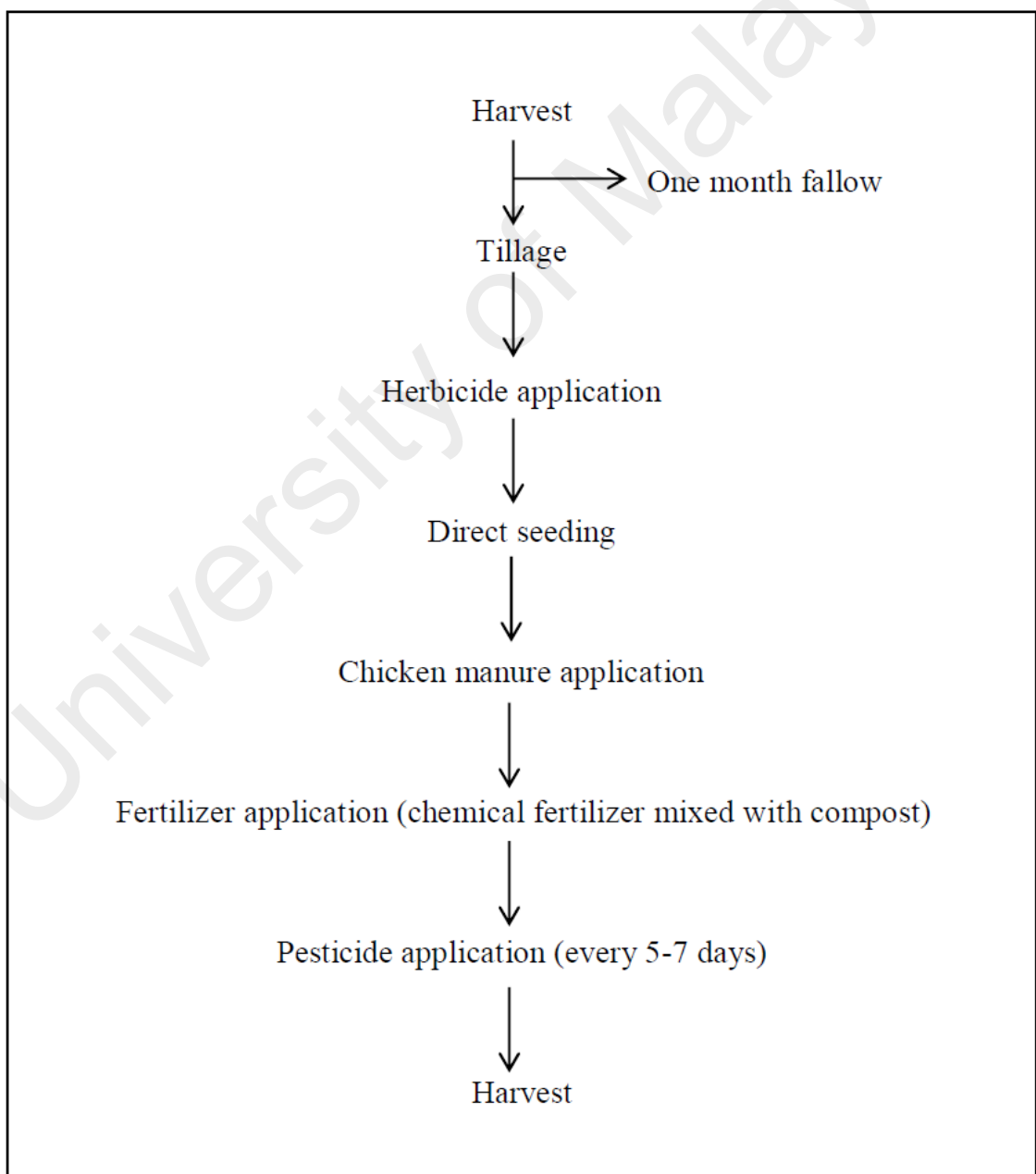


Figure 3.9: Farming process at CF1

Figure 3.10 shows the farming process at CF2 which after crop harvesting the soil was only allowed to rest for one day before next cropping. Herbicide was applied right after harvest and the soil was tilled after the one day fallow. Chicken manure was applied immediately after seeding and chemical fertilizer was applied one week after seeding. Pesticide application was done every 3-7 days depending on climate and pest incident.

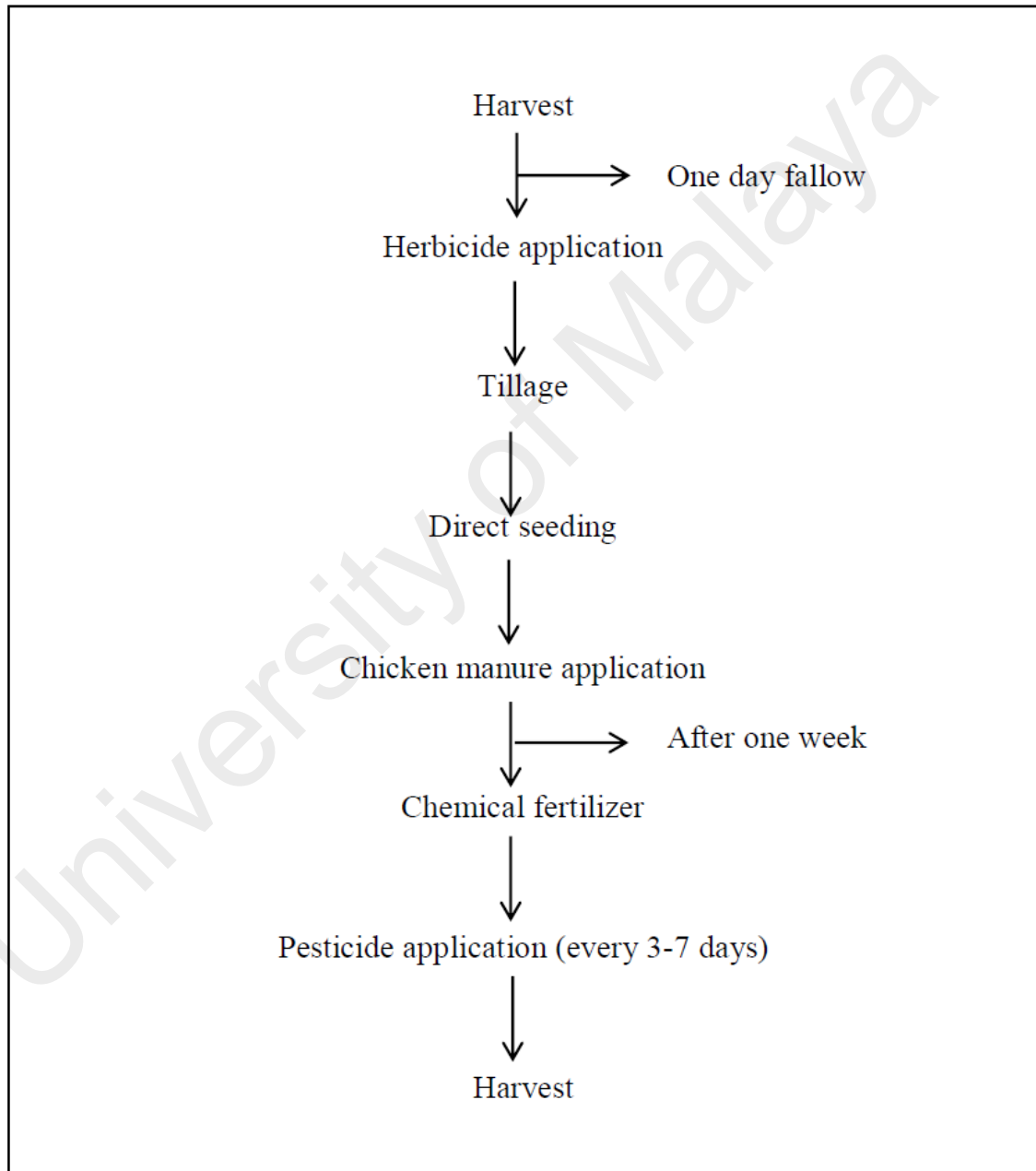


Figure 3.10: Farming process at CF2

Figure 3.11 shows the farming process at OF1. After each harvesting, the soil was allowed to fallow for two weeks. Before seedling transplanting, the soil was tilled and compost was applied as basal fertilizer. After transplant of seedling, Bokashi compost was applied. Pest repellent was applied weekly for pest control pest.

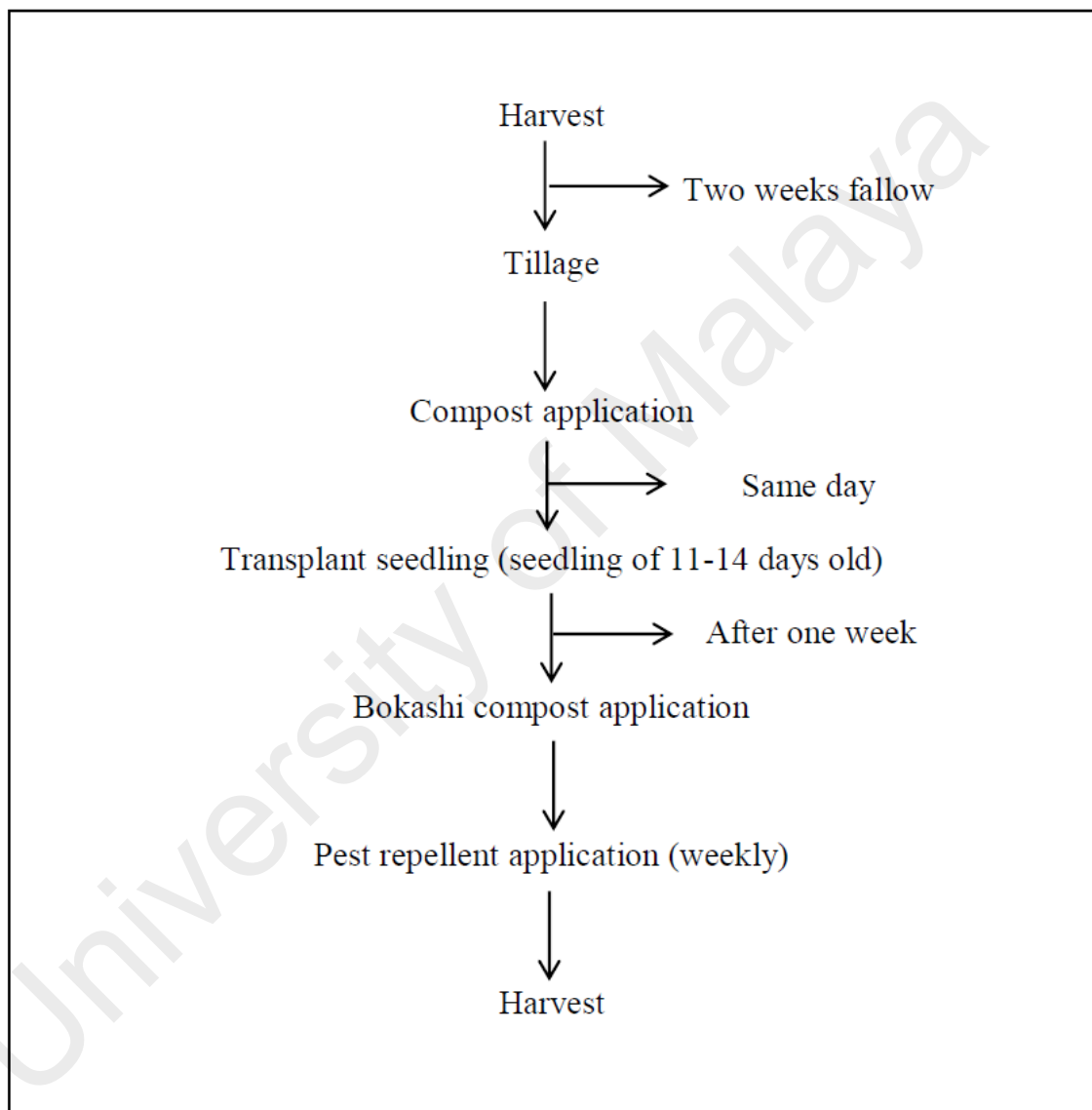


Figure 3.11: Farming process at OF1

Figure 3.12 shows the farming process at OF2. The soil was allowed to rest and fallow for two weeks after harvest. After tillage, gypsum was applied right before transplanting

of seedling. One week after transplanting, compost was applied. Pest repellent was applied 2-3 times per cropping cycles.

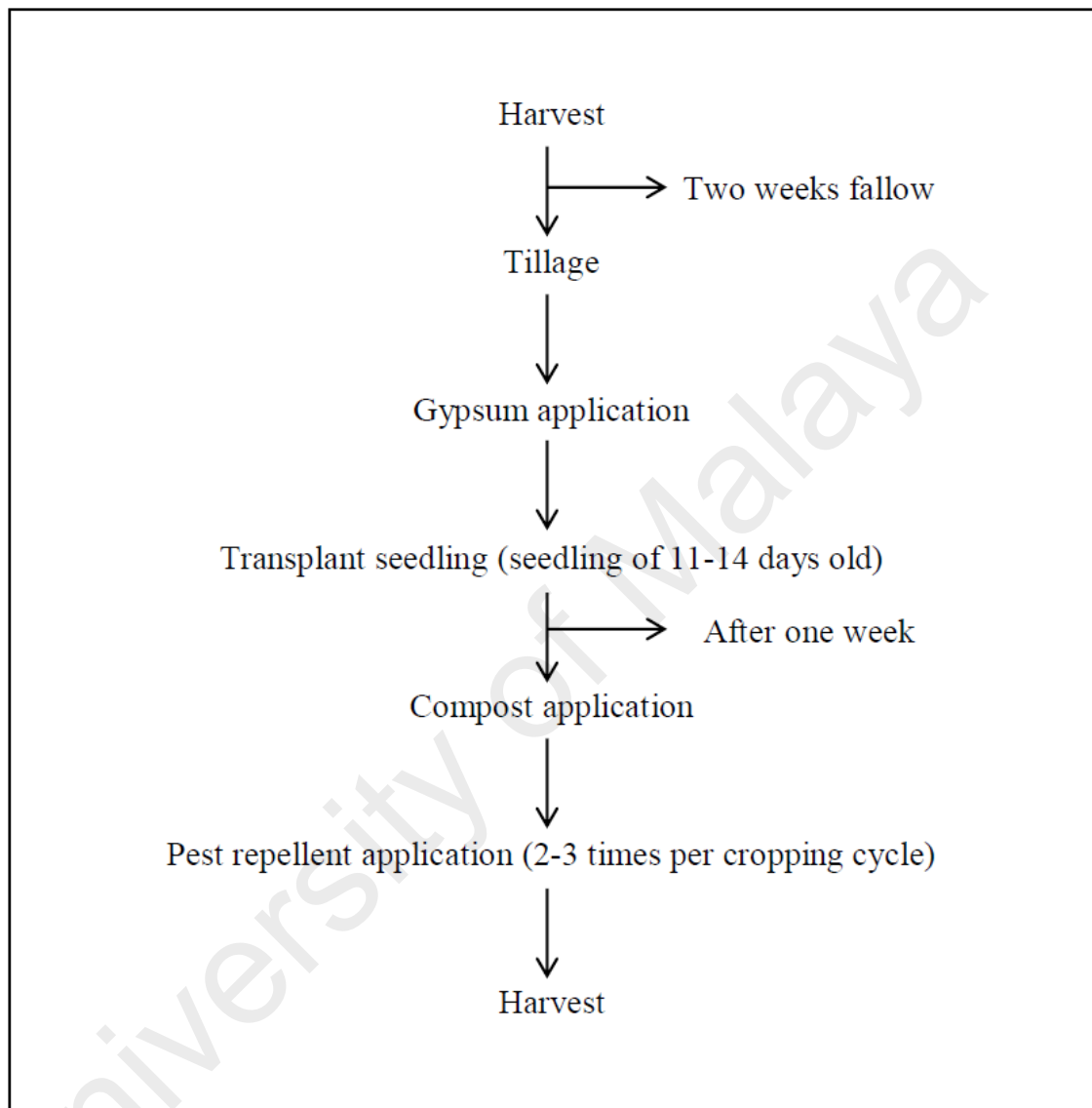


Figure 3.12: Farming process at OF2

The farms practiced intensive year-round production, which involved daily irrigation, except for CF1 and CF2, the fields are irrigated only on dry days. The farms also practiced crop rotation. About 6, 11, 8, and 8 cropping cycles are annually attainable per plot at CF1, CF2, OF1, and OF2, respectively. The market-oriented farms produced different varieties of leafy vegetables as listed in Table 3.7.

Table 3.7: List of vegetable variety grown in farm

Local Name	English Name	Scientific Name	Farm Types <sup>a</sup>			
			CF1	CF2	OF1	OF2
Amaranth	Amaranth	<i>Amaranthus spp.</i>	√	√	√	√
Amaranth, Red	Amaranth, red stripe leaf	<i>Amaranthus spp.</i>			√	√
Choy Sum	Chinese flowering cabbage, chinese soup green, white flowering cabbage, mock pak choy, choy sum	<i>Brassica rapa var. parachinensis</i>	√	√		√
Fu Gui Choy	Chinese soup green, white flowering cabbage, mock pak choy, choy sum	<i>Brassica rapa var. parachinensis</i>			√	
Fu Mak, tong ho	Garland chrysanthemum, chrysanthemum greens, edible chrysanthemum	<i>Chrysanthemum coronarium</i>			√	
Hong Kong Choy Sum	Chinese flowering cabbage, chinese soup green, white flowering cabbage, mock pak choy, choy sum	<i>Brassica rapa var. parachinensis</i>	√		√	
Japanese Choy Sum	Chinese flowering cabbage, chinese soup green, white flowering cabbage, mock pak choy, choy sum	<i>Brassica rapa var. parachinensis</i>		√	√	
Kai Lan	Chinese kale, white flowering broccoli, kailan	<i>Brassica oleracea var. alboglabra</i>			√	
Lettuce	Vietnamese lettuce, chinese lettuce, leaf lettuce, curled lettuce	<i>Lactuca sativa var. crispa</i>			√	√
Mini Cos Lettuce	Romaine lettuce	<i>Lactuca sativa L. var. longifolia</i>			√	

Source: Vujovic & Lorimer (2009)

<sup>a</sup> CF1 conventional farm located at central region Malaysia, CF2 conventional farm located at southern region Malaysia, OF1 organic farm located at central region Malaysia, OF2 organic farm located at southern region Malaysia

Table 3.7: List of vegetable variety grown in farm (Continued)

Local Name	English Name	Scientific Name	Farm Types <sup>a</sup>			
			CF1	CF2	OF1	OF2
Nai Bai	Bukchoy, chinese chard, chinese white cabbage, mustard cabbage, baicai, pakchoy, bokchoy	<i>Brassica rapa subsp. chinensis</i>			√	√
New Zealand Spinach	New zealand spinach, sea spinach, botany bay spinach, tetragon and cook's cabbage	<i>Tetragonia tetragonioides</i>			√	√
Ong King Pak Choy	Baby chinese chard, chinese white cabbage, chinese mustard, celery mustard	<i>Brassica rapa var. chinensis</i>			√	
Senposai	Senposai is a new hybrid of cabbage and Komatsuna	<i>Brassica rapapekinensis x brassica campestris</i>			√	
Sweet Potato Leaf	Sweet Potato Leaves	<i>Ipomea batatas</i>		√		
Water Spinach	Water convolvulus, tropical spinach, water ipomea, water spinach, water sweet potato, swamp cabbage, swamp morning glory, kangkong	<i>Ipomea aquatica, I. reptans</i>			√	√
Xiao Pak Choy	Shanghai chinese chard, shanghai chinese chard	<i>Brassica rapa var. chinensis</i>	√	√	√	√
Xiu Zhen Choy Sum	Chinese soup green, white flowering cabbage, mock pakchoy, choy sum	<i>Brassica rapa var. parachinensis</i>		√	√	√

Source: Vujovic & Lorimer (2009)

<sup>a</sup> CF1 conventional farm located at central region Malaysia, CF2 conventional farm located at southern region Malaysia, OF1 organic farm located at central region Malaysia, OF2 organic farm located at southern region Malaysia



### 3.2.3 Key stock, process, and flow identification

Data was collected over a period of 24 months using an integrated method of desk research, field visit, site observation, and personal interview with the farmers and personnel from Department of Agriculture, Malaysia. Farm activities were investigated and characterized to identify the main processes, stocks, and flow within system boundaries. The basic descriptions recorded farm location, average annual precipitation and temperature, soil types, cultural practices, arable area, yield, crop rotation system, monthly fertilizer input and variety of crops. The key input and output were identified during data collection. Monthly usage of compost, chicken manure, pest repellent, seeds, herbicide, pesticide, chemical fertilizer and vegetable production data were provided by farm owner.

The key materials associated with farm inputs–outputs were identified and quantified. Eight inputs (Bokashi compost, compost, vermicompost, peat moss, chicken manure, chemical fertilizer, rainfall, irrigation water) and five outputs (harvested crop, surface runoff and leaching, gaseous emission, waste water, and organic waste) associated with C and N flow at the farms were identified. The volume or tonnage of input-output material data were collected from interview with farmers or Department of Agriculture Malaysia. Soil, vegetable and water samples were collected from farms for chemical analysis to determine C and N concentrations according USDA standard (Section 3.4 and 3.5). C and N fluxes were estimated by multiplying the mass of inputs–outputs with the C and N concentrations according to Eq. 3.2 and 3.3 (Cencic & Rechberger, 2008; Prasad & Hochmuth, 2014; Prasad *et al.*, 2015).

For solid input-output:

$$TM_i = M_i \times C_i \quad \text{Eq. 3.2}$$

where,  $TM_i$ , total mass of C or N of material  $i$  (kg ha<sup>-1</sup>);  $M_i$ , total mass of material  $i$  (kg dw ha<sup>-1</sup>);  $C_i$ , C or N concentration of material  $i$  (%).

For water input-output

$$TM (\text{Water}) = V \times C_i \times 0.01 \quad \text{Eq. 3.3}$$

where,  $TM (\text{Water})$ , is the total mass of C or N in water (kg ha<sup>-1</sup>);  $V$ , is total of water applied (mm);  $C_i$  is total C or N concentration (mgL<sup>-1</sup>); 0.01, converting mgL<sup>-1</sup> and cubic meter per mm to kg ha<sup>-1</sup>.

### 3.2.4 Mass balance analysis

The farms were categorized either as sink, equilibrium or source based on the mass balance generated from STAN model (Eq. 3.4). The soil stock was calculated to provide fundamental information for C and N flux modeling (Eq. 3.4) (Zubrzycki *et al.*, 2013).

The C and N stock was then incorporated into the STAN modeling to indicate the current C and N stock during study.

$$\Delta, farm = \int_{t_0}^t m, input (\tau) d\tau - \int_{t_0}^t m, output (\tau) d\tau = \begin{cases} > 0 : sink \\ 0 : equilibrium state \\ < 0 : source \end{cases}$$

Eq. 3.4

where,  $\Delta$  is the change in mass over period of time,  $\int_{t_0}^t m, input (\tau)d\tau$  is the input flux rate while  $\int_{t_0}^t m, output (\tau)d\tau$  is the output flux rate.

$$S = BD * PD * C * 10000 \quad \text{Eq. 3.5}$$

where, S is soil C or N stock ( $\text{kg ha}^{-1} \text{ y}^{-1}$ ); BD, bulk density ( $\text{kg m}^{-3}$ ); PD is the plough depth (0.1 m); C, soil C or N concentration (%); 10000, convert to per hectare basis

### 3.2.5 Modeling

STAN 2.5 (subSTance flow ANalysis) by inka software<sup>®</sup> is a software program that performs a two-layer analysis: MFA and SFA. Material flow includes the mass of materials that enter or exit the system, such as compost, fertilizer, and crops. SFA examines element fluxes in chemical terms (C and N in this study). The material flow layer provides a structural design for the substance flow layer, which links the material with N balance. In this study, graphical model of material flow, C flow and N flow were constructed on the basis of predefined processes, flows, subsystems, and coefficients with STAN 2.5 (Cencic & Rechberger, 2008).

### 3.3 Field Sampling

Soil, vegetable, compost, fertilizer, manure, organic waste and water were sampled from the field through random composite sampling method and analysed in the laboratory (January 2012 to December 2014.). Composite samples of vegetables, compost, fertilizer, and manure were collected through sub-sampling ( $n = 15$ ) for each load or each harvest time. Soil samples were collected for six times (average every six months). Each composite soil sample consisted of 15 cores (diameter of 5 cm, depths of 0–15 and 15–30 cm) taken in diagonal patterns across each plot (EPA, 2012). Collected samples were packed into clean and sterile plastic bag and analyzed within 48 hours.

Leachate was collected in triplicate with a soil water sampler at a soil depth of 25 cm at the beginning of each planting cycle and water sample was collected with peristaltic pump for physico-chemical analysis. Leachate samples of three planting cycles were collected for analysis (Voll & Roots, 1999). Wooden sticks were used as marking for installed soil water sampler and the two protrude polyvinyl tubes were tied onto the stick and covered with plastic cover to prevent any foreign objects from entering. The soil water samplers were constructed based on Migliaccio et al.(2006) with materials below (Plate 3.5). This includes:

- 5-gallon container to store water samples that flow through the collection hole,
- Container lid covers the container to ensure no foreign material contaminate the sample,
- Connector for polyvinyl tubing (2pcs),
- Polyvinyl tubes: sample collection and air vent (2pcs is 50cm in length), sample collection tube (1 pcs is 10cm in length),
- Collection plate was glued on top of the container lid with the collection hole align together,
- Mesh filter was glued onto the collection hole of the collection plate to filtered out soil, rocks and other debris,
- Acid-washed sand placed on top of collection plate to prevent clogging of the mesh filter,
- Silicone sealant to ensure no crack when the connector fit onto the container, and
- Driller was used to drill hole for water sample collection on the container lid and the collection plate. Two holes were drilled at the side of the container to fit the connector (Migliaccio *et al.*, 2006).



Plate 3.5: Soil water sampler

A sub-sample of surface runoff was collected at every output point while rain water samples were collected with a rain gauge after each rain event. Sub-sample of surface runoff was collected for three times from each output point for composite sample analysis. Rain water samples were collected and analysed three times with rain gauge. The water samples collected were stored with acid-washed, sterile plastic bottles and analysed within 24 hour.

### **3.4 Solid samples analysis**

All solid samples (soil, manure, compost, fertilizer, vegetables and peatmoss) were dried at 80 °C, ground and sieved (1 mm mesh) prior to analysis with an elemental analyzer analyser (Perkin Elmer CHNS/O Series II 2400) (Elmer, 2010). Sample weights of 1-2mg were measured with PerkinElmer AD6 Autobalance (Culmo & Shelton, 2013). The samples were packed into tin capsules (PerkinElmer, N2411362) and analysed for C and N content based on Pregl-Dumas method with fully automatic elemental analyser (Pereira *et al.*, 2006; Wendling *et al.*, 2010). Total C and total N was analyzed using CHNS Elemental Analyzer which based on Pregl-Dumas method where samples were combusted in a pure oxygen environment (Wendling *et al.*, 2010). Analysis of K and P for soil sample follows the standard method of ASTM E 926-94 (Total Potassium) and ASTM D 5198-92 (Total Phosphorous).

#### **3.4.1 Heavy Metal Analysis**

Prior to analysis the solid samples were digested with acid based on USEPA Method 3050B (EPA, 1996). One gram of soil or sediments was digested by mixing with 10ml of concentrated HNO<sub>3</sub> and heated at 95°C ± 5°C under reflux for 10 to 15 minutes. The sample was allowed to cool and 5ml of concentrated HNO<sub>3</sub> was added and refluxed for another 30 minutes. 5ml of HNO<sub>3</sub> was added and refluxed until no brown fumes were released from the sample. When brown fumes are not seen, the sample was cooled and 2ml of water and 3 ml of 30% H<sub>2</sub>O<sub>2</sub> were added and refluxed again. 1 ml of 30% H<sub>2</sub>O<sub>2</sub> was added when effervescence was minimal. The sample was continuously heated without boiling until the volume reduced to approximately 5ml. The sample was then diluted with 100ml of deionized water and filtered (Whatman No. 1) before the sample was sent for analysis. The samples were analyzed using inductive coupled plasma mass spectrometry (ICP-MS) for metal element (Cadmium, Calcium, Copper, Lead,

Magnesium, Manganese) analysis based on standard method using USEPA Method 6020A (EPA, 2007).

### **3.5 Physical and chemical analysis of water samples**

Water analysis assesses the physico-chemical parameters, in order to identify any changes or possibilities in water pollution distress. Physical, biological and chemical parameters were tested to provide the background of water quality for better understanding on the relationship between the element flow and water quality. Water samples were collected and analyzed with a spectrophotometer (HACH DR/4000), Eutech Instruments EcoScan DO<sub>6</sub>, and Eutech Instruments CyberScan con11. The water samples collected were stored at 4°C and analysed within 48 hours. The water samples were analysed for various physical, biological and chemical parameters (Table 3.8).

#### **3.5.1 Water flow**

Input and output volume of irrigation water, water usage, waste water, runoff, and leachate were obtained through field observation, direct measurement and estimation. The volume of irrigation water and runoff was measured with flow meter at every water output point in the farm (Eq. 3.6) (Boman & Shukla, 2009). Equation 3.7 estimated the total water flow in the farms (Bengtsson *et al.*, 2003). The conversion of rainfall and evapotranspiration rate to volume per unit area was done according to FAO methods (Eq. 3.8 and 3.9) (Brouwer, 1998).

$$\text{Water volume} = \text{flow rate (cm per min)} \times \text{duration of flow (min)} \quad \text{Eq. 3.6}$$

Table 3.8: Physical, biological and chemical analysis of water samples

<b>Parameters</b>	<b>Equipment</b>	<b>Method</b>
pH	Mettler Toledo pH Meter	Direct measurement
Total Dissolved Solid	Eutech Instruments CyberScan con11	Direct Measurement in field
Conductivity	Eutech Instruments CyberScan con11	Direct Measurement in field
Temperature	Eutech Instruments CyberScan con11	Direct Measurement in field
Dissolved Oxygen (DO)	Eutech Instruments EcoScan Do6	Direct measurement
Biochemical Oxygen Demand (BOD <sub>5</sub> )	Eutech Instruments EcoScan Do6	Direct Measurement (APHA, 1998)
Chemical Oxygen Demand (COD)	HACH Spectrophotometer DR/4000	HACH Procedure Method 8000 Reactor Digestion Method (APHA, 1998)
Total Suspended Solid	HACH Spectrophotometer DR/4000	Direct measurement
Hardness	HACH Spectrophotometer DR/4000	HACH Procedure Method 8030 Calcium and Magnesium; Calmagnite Colorimetric Method
Chloride	HACH Spectrophotometer DR/4000	HACH Procedure Method 8113 Mercuric Thiocyanate Method
Turbidity	HACH Spectrophotometer DR/4000	HACH Procedure Method 10047 Attenuated Radiation Method )Direct Reading)
Total Organic C	HACH Spectrophotometer DR/4000	HACH Procedure Method 10128 Direct Method
Total N	HACH Spectrophotometer DR/4000	HACH Procedure Method 10072 Persulfate Digestion Method
Total Inorganic N	HACH Spectrophotometer DR/4000	HACH Procedure Method 10021 Titanium Trichloride Reduction Method
Ammonia	HACH Spectrophotometer DR/4000	HACH Procedure Method 10031 Salicylate Method
Nitrite	HACH Spectrophotometer DR/4000	HACH Procedure Method 8153 Ferrous Sulfate Method
Nitrate	HACH Spectrophotometer DR/4000	HACH Procedure Method 8039 Cadmium Reduction Method



Water outflow (surface runoff and leaching) = Total water inflow (precipitation and irrigation) - evapotranspiration

Eq. 3.7

Total rainfall ( $\text{m}^3 \text{ ha}^{-1} \text{ year}^{-1}$ ) = total rainfall ( $\text{mm year}^{-1}$ ) x 1 m  $\div$  1000 mm x 10000  $\text{m}^2$

Eq. 3.8

Total evaporation ( $\text{m}^3 \text{ ha}^{-1} \text{ year}^{-1}$ ) = total evaporation ( $\text{mm year}^{-1}$ ) x 1 m  $\div$  1000 mm x 10000  $\text{m}^2$

Eq. 3.9

Conventional farms are open field planting, thus, no water irrigation was applied during raining days. On the other hand, rainfall do not affect irrigation regime at organic farms where crops are cultivated under rain shelter. The mean monthly evapotranspiration for open field planting ranged from 1,360  $\text{mm y}^{-1}$  to 1,490  $\text{mm y}^{-1}$  (Abdullahi *et al.*, 2013a; Abdullahi *et al.*, 2013b; Ali *et al.*, 2000; Ali & Shui, 2009; Arshad, 2014; Kumagai *et al.*, 2005; Kume *et al.*, 2011; Lee *et al.*, 2004; Tukimat *et al.*, 2012). The evapotranspiration under all field sheet covers tend to be lower than that under open field by 2.0–2.5  $\text{mm d}^{-1}$  (Hashem *et al.*, 2011). The daily rainfall data during the study period were obtained from a meteorological station nearest to the study farms (50km radius). Below is the meteorological station:

- Station: Ampangan Air Sungai Semenyih (Latitude: 02° 56' N, Longitude: 101° 52' E)
- Station : Pusat Latihan Pertanian Kalumpang (Latitude: 03° 38' N, Longitude: 101° 29' E)
- Station : Subang (Latitude: 03 ° 07 ' N, Longitude: 101 ° 33 ' E)

### 3.6 Biological fixation

Biological fixation is one of the important N inputs in agriculture soil. The non-symbiotic N fixation in this study was estimated according to Eq. 3.10 (Abdulkadir *et al.*, 2013).

$$N \text{ fixed} = 0.5 + 0.1P^{\left(\frac{1}{2}\right)} \quad \text{Eq. 3.10}$$

where N fixed, is the non-symbiotic N fixation, kg ha<sup>-1</sup>; P, is the total precipitation volume (mm y<sup>-1</sup>)

### 3.7 Total C derived from photosynthesis

Total C derived from photosynthesis is defined as crop residues, root material and exudates left on soil after cultivation which is an important C input in agriculture soil. The total C derived from photosynthesis (above ground) was calculated based on equation below (Eq. 3.11):

$$\text{Total C derived from photosynthesis (above ground), kg} = P_i \times (1 - W_i\%) \times C_i\% \quad \text{Eq. 3.11}$$

where, P<sub>i</sub> is the vegetable production of farm i, kg; W<sub>i</sub>% and C<sub>i</sub>% are the water content and the C content of harvested vegetable at farm i, respectively.

Total C derived from photosynthesis was estimated by multiplying total C from harvested vegetable by a factor of 1.4 by assuming that root dry matter and net exudation constituted an average of 30% of total assimilated C (Goenster *et al.*, 2014;

Kuzyakov & Domanski, 2000; Safi *et al.*, 2011b). Therefore, the C derived from photosynthesis (roots and exudates) below ground was estimated by Eq. 3.12:

$$\text{C derived from photosynthesis} = \text{Total C derived from photosynthesis} - \text{Total C derived from photosynthesis (above ground)} \quad \text{Eq. 3.12}$$

### 3.8 Gaseous emission

The emission of gaseous CO<sub>2</sub>, CO, NH<sub>3</sub>, and CH<sub>4</sub> emitted directly from agriculture soil were measured by determining the rate of gas concentration change in the headspace of static chamber (32 cm × 22 cm × 22 cm) using a portable gas meter (Binder Combimass GA-m multi-element) (Parkin & Venterea, 2010; Rochette & Gregorich, 1998). The static chamber was made of non-reactive materials acrylic, also known as plexi-glass (2mm thick) coated with aluminium foil. One outlet was installed at the top of the chamber that was fitted with silicone tube (5cm length) (Figure 3.13). The end of silicone the tube was closed with butyl rubber septa for sampling. A fan (Ultimax BW-6025D12, 12V 0.2A) was installed in the chamber to mix headspace for homogenous sampling. Gaseous emission was measured every 5 minute for one hour with portable gas meter (Binder Combimass GA-m multi-element), with the sampling tube installed with needle. The needle was easily inserted through the butyl rubber septa of the chamber outlet, which allowed the portable gas meter to measure the gases in the chamber.

Gaseous emissions were measured weekly for a crop cycle with three replicate chambers randomly installed on the day of measurement to avoid any disturbance to the farms' daily activities. Gaseous flux was measured every 6 months and all measurements were taken between 9 a.m and 12 noon of the day. The chamber was not

placed longer than 60 min for each measurement to reduce variability in gaseous flux and chamber, hence reducing induced biases.

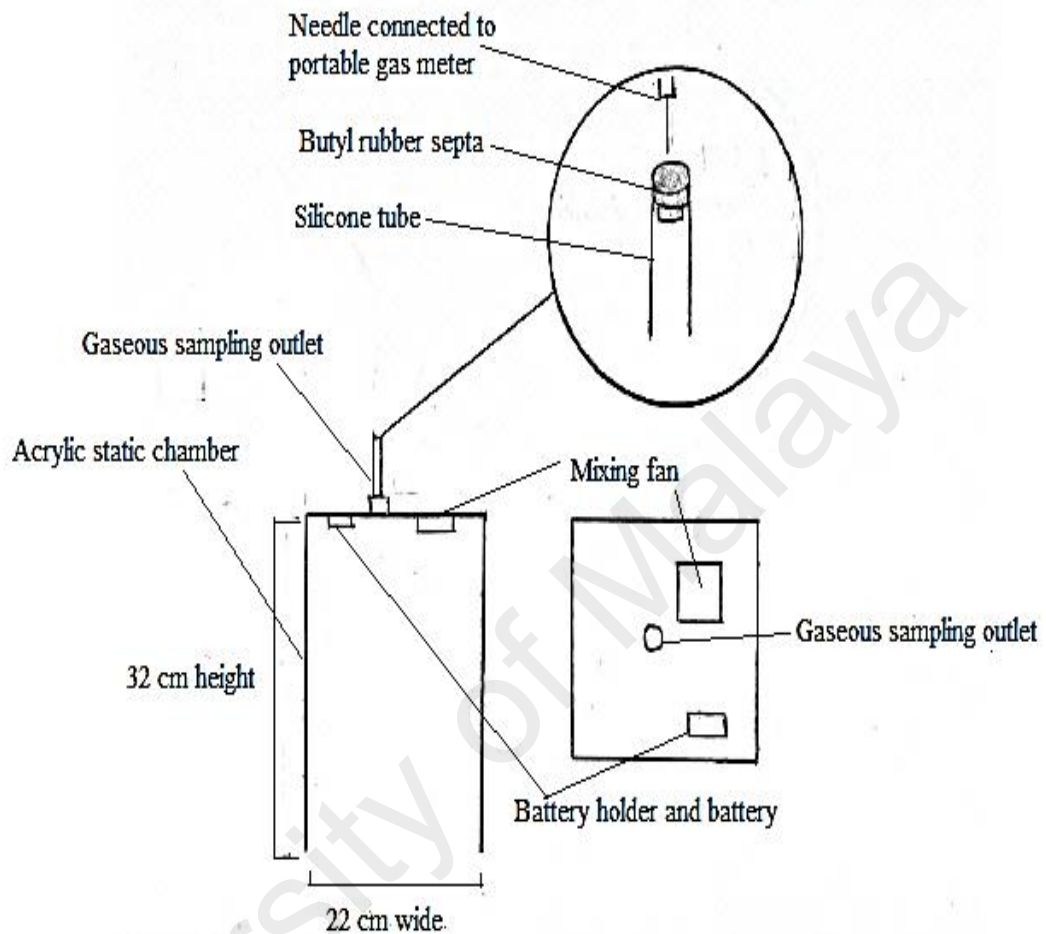


Figure 3.13: Static chamber layout used for gas sampling

### 3.8.1 Gaseous flux rate calculation

Gaseous emission readings from portable gas meter were measured in part per million (ppm). Therefore to calculate rate of gaseous emission, the standard curve was plotted where x-axis was time (min) and y axis was gas concentration (ppm). The slope of the graph is the flux rate of that particular gas with unit of  $\mu\text{L gas L}^{-1} \text{ min}^{-1}$ . The C and N mass escaping from the farm system in gaseous form were calculated based on ideal gas law with the gas flux rate reading from the gas chamber measurement. The gas flux and

total C and N mass were calculated according to the Eq. 3.13 and 3.14, respectively (Parkin & Venterea, 2010).

$$GF_i = \left( \frac{y_2 - y_1}{x_2 - x_1} \right)_i * \frac{VC}{AC} \quad \text{Eq. 3.13}$$

where,  $GF_i$  is the gas flux volume of gas  $i$  ( $\mu\text{L gas L}^{-1} \text{ m}^{-2} \text{ min}^{-1}$ );  $\left( \frac{y_2 - y_1}{x_2 - x_1} \right)_i$  is the slope of standard curve graph of gas  $i$  ( $\mu\text{L gas L}^{-1} \text{ min}^{-1}$ ); VC is the volume of gas chamber (L); AC, surface area of gas chamber ( $\text{m}^2$ )

$$TM_i = \left( \frac{PVi}{RT} \right) * \left( \frac{1}{10^{-6}} \right) * M * 10000 \quad \text{Eq. 3.14}$$

where,  $TM_i$  is the total C or N mass of gas  $i$  ( $\text{g min}^{-1}$ ); P is the atmosphere pressure based on farm altitude (atm);  $V_i$  is the gas flux volume of gas  $i$  obtained from equation (8) ( $\mu\text{L gas L}^{-1} \text{ min}^{-1}$ ); R is the gas law constant (0.08206); T is the temperature ( $\text{K}^\circ$ );  $1/10^{-6}$  is the unit conversion from micro-mole to mole; M is the C ( $12.0107 \text{ g mol}^{-1}$ ) or N ( $14.007 \text{ g mol}^{-1}$ ) molar mass; 10000 is required to express the results in the per hectare .

Nitrous oxide ( $\text{N}_2\text{O}$ ) emission was calculated according to IPCC calculation (Eq. 3.165 (Bouwman, 1996; Freney, 1997; Sims *et al.*, 2007; Smith *et al.*, 2001). The N input from atmospheric dry deposition was assumed to be negligible (Jackson *et al.*, 2003).

$$E_{\text{N}_2\text{O}} = 1 + 0.0125 \times N_{\text{input}} \quad \text{Eq. 3.15}$$

where  $E_{\text{N}_2\text{O}}$  is the emission rate of nitrous oxide ( $\text{kg N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ ); the value of 1 is the background emission rate assumption;  $N_{\text{input}}$  is the total N input ( $\text{kg N ha}^{-1} \text{ y}^{-1}$ ).

The Mass of CO<sub>2</sub>E of each farm systems were calculated by multiplying the mass of gas with GWP (Table 3.9) based on IPCC method (Eq. 3.16) (IPCC, 2007, 2014a)

$$\text{Mass of CO}_2\text{E} = \text{mass of gas} \times \text{GWP} \quad \text{Eq. 3.16}$$

Table 3.9: GWP of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O

Gaseous	Lifetime (years)	GWP time horizon	
		20 years	100 years
CO <sub>2</sub>	1	1	1
CH <sub>4</sub>	12.4	86	34
N <sub>2</sub> O	121.0	4950	7350

### 3.9 Microbial plate count

The soil, vegetable and water samples collected from the farms were stored in a sterilized container. The samples were transported in a chest cooler with ice pack that maintained the sample at 0-4°C until arrival at the laboratory (Andrews & Hammack, 2001). The samples were analyzed within 2 hour after sample collection. The collected vegetable samples were washed with distilled water to remove any soil residues. Only the edible parts of the vegetables were ground with sterilized pestle and mortar. 1g or 1ml of samples (plant, soil and water) were transferred into sterilized test tube with 9 ml of distilled water and mixed with vortex mixer. One ml of diluted sample was pipetted and spreaded onto triplicate plates of three types of agar: nutrient agar-Difco (general microbial plate count), MacConkey Agar-Difco (*E. coli*/coliform plate count) and SS agar-Difco (*Salmonella/shigella* plate count) and incubated for 48 hour at 35°C (Andrews & Jacobson, 2001; Andrews *et al.*, 2001; Feng *et al.*, 2002; Keller, 2014). The agars were prepared according to manufacturer's instructions (Difco™ Manual). The bacterial colonies formed were counted and recorded as colony forming unit per gram of fresh sample (CFU g<sup>-1</sup>) according to US FDA methodology (Maturin & Peeler,

2001). The genus of the isolated bacteria were identified with Gram staining and biochemical tests according to Bergey's manual (Bergey *et al.*, 1994):

### 3.9.1 Identification of Bacterial Isolates

The bacteria isolated were identified based on Bergey's Manual using biochemical characterization) (Bergey *et al.*, 1994). Reagents and agars used for biochemical test were prepared according to manufacturer's instruction (Difco & BBL Manual Manual of Microbiological Culture Media) (Zimbrow *et al.*, 2009).

### 3.10 Farm management efficiency

Farm efficiency in terms of fertilizer used, water used, C input, N used, waste generation, farm production, and cost profit were evaluated (Table 3.10) (Hřebíček *et al.*, 2013). Fraction of N use efficiency and N loss were calculated based on Eq. 3.17 and 3.18. The total N loss used in the calculation was based on the total N output estimated in N flow analysis using STAN 2.5 software.

Table 3.10: Farm efficiency indicators and definition

<b>Indicator</b>	<b>Definition</b>
Fertilizer use efficiency	Fertilizer use per crop production, $\text{kg kg}^{-1}$
Water use efficiency	Water use per crop production, $\text{m}^3 \text{kg}^{-1}$
C input efficiency	C input per crop production, $\text{kg C kg}^{-1}$
N use efficiency	N use per crop production, $\text{kg N kg}^{-1}$
Waste generate rate	Waste produce per production, $\text{kg kg}^{-1}$
Production	Yield per area of land use, $\text{kg ha}^{-1}$

$$\text{N use efficiency fraction} = \text{Total vegetable production} / \text{Total N input} \quad \text{Eq. 3.17}$$

$$\text{N loss fraction} = \text{Total N loss} / \text{Total vegetable production} \quad \text{Eq. 3.18}$$

### 3.11 Cost volume profit (CVP) analysis

The cost volume profit (CVP) analysis examined the basic managerial accounting info of sales volumes, costs and prices (Weygandt *et al.*, 2012) which shows the relationship between farm revenue and operational costs. The CVP of each study farm was assessed based on Eq. 3.19-3.25 (Garrison *et al.*, 2011; Noreen *et al.*, 2011):

$$OP = TR - TC \quad \text{Eq. 3.19}$$

where, OP, is operating profit; TR, is total revenue; TC, is total costs

$$TR = P \times Q \quad \text{Eq. 3.20}$$

where, TR, is total revenue; P, is selling price per unit; Q, is quantity of produce (vegetable yield)

$$TC = C + D + W + M \quad \text{Eq. 3.21}$$

where, TC, is total costs; C, is total cost for consumable (pesticide, herbicide, fertilizer, chicken manure, compost material, seeds, and peat moss); D, is total cost for diesel usage; W, is total cost for wages; M, is total cost for farm maintenance

$$CM = TR - TC \quad \text{Eq. 3.22}$$

where, CM, is contribution margin; TR, is total revenue; TC, is total costs

$$CU = TC / Q \quad \text{Eq. 3.23}$$



where, CU, is cost per unit; TC, is total costs; Q, is quantity of produce (vegetable yield)

$$CMU = CM/Q \quad \text{Eq. 3.24}$$

where, CMU, is contribution margin per unit; CM, is contribution margin; Q, is quantity of produce (vegetable yield)

$$CMR = CM/TR \quad \text{Eq. 3.25}$$

where, CMR, is contribution margin ratio; CM, is contribution margin; TR, is total revenue

University of Malaya

## CHAPTER 4: RESULTS AND DISCUSSION

### 4.1 Characterization of conventional and organic vegetable farm

Based on the farm operation, management and certification, CF1 and CF2 are classified as conventional system while OF1 and OF2 are classified as organic system. The selected organic farms have been certified by the Malaysia organic certification SOM. According to the date when the farms were established, CF2 and OF1 are considered young farms with three to three and a half years of establishment as compared to CF1 and OF2 which have been operating for nine to ten years, respectively. Both OF1 and OF2 cultivate vegetables under rain shelter, while conventional farms practiced open field planting. All the farms in this study practiced intensive irrigation with sprinkler system. However, the open field planting practices in CF1 and CF2 permit rainfall to enter the farm soil directly to complements the sprinkler irrigation system. In general, the crop varieties in the organic system were more diversified than the conventional system in order to increase farm biodiversity.

#### 4.1.1 Soil properties

The physical-chemical soil properties differed among the farms in this study that may influence the elements which flow into the soil. Table 4.1 shows the physical-chemical properties of the soil samples collected from the farms in this study. The N level in soil samples from conventional farms was twice higher as compared to the organic farms. The classifications of N levels are: very low (<0.05%), low (0.05– 0.15%), medium (0.15– 0.25%), high (0.25–0.50%), very high (>0.5%) (APAL, 2015). The soil N level in CF1 (0.292%) and CF2 (0.574%) were classified as high and very high while OF1 (0.162) and OF2 (0.217) were recorded as medium soil N level. The soil C of the farms in this study were in the range of 2-5%. Soil C is important for soil fertility and it is

applied into soil through soil amendments such as crop residues, composts, and manures (Mobar *et al.*, 2015). The soil N and C concentrations in this study were generally higher than those reported by Zhang *et al.* (2011) in wheat-maize rotation soil, ranging between 0.064-0.107% and 0.57-0.8%, respectively. The differences might be due to the variation of crop types and fertilizer regime (Chen *et al.*, 2004; Simonne & Hochmuth, 2005).

Table 4.1: Physical-chemical properties of soil samples in farms in this study

	CF1	CF2	OF1	OF2
<b>Soil pH</b>	6.9-7.8	6.0-7.1	7.2-7.5	7.3-7.5
<b>Soil C:N ratio</b>	10:1	7:1	15:1	13:1
<b>Soil bulk density (kg m<sup>-3</sup>)</b>	595	545	422	467
<b>Organic matter (%)</b>	3.7	7.8	5.8	3.6
<b>Total N (%)</b>	0.292	0.574	0.162	0.217
<b>Total C (%)</b>	2.16	4.51	3.38	2.09
<b>Total Potassium (mg/kg)</b>	430	524	1,135	875
<b>Total Phosphorous, (mg/kg)</b>	1,957	1,051	1,674	1,247
<b>Magnesium (mg/kg)</b>	2,042	374	1,579	818
<b>Calcium (mg/kg)</b>	12,175	1,535	13,040	4,630
<b>Manganese (mg/kg)</b>	101	62.8	203	67.8
<b>Copper (mg/kg)</b>	23.5	18.9	22.6	9.72
<b>Lead (mg/kg)</b>	ND (<0.01)	ND (<0.01)	52.3	ND (<0.01)
<b>Cadmium (mg/kg)</b>	ND (<0.01)	ND (<0.01)	ND (<0.01)	ND (<0.01)

ND – Not detectable

The soil C:N can be classified as very low (<8), low (8-10), medium (10-15), high (15-25), and very high (>25) (Hill, 2015). The C:N of top soil (20cm) in the farms in this study were generally within the medium level except for CF2 which has C:N ratio of 7 that was classified as very low. The soil C:N indicated the soil fertility and the interactions between soil C and N which were often influenced by climate, soil conditions, vegetation types, and agricultural managements (Lou *et al.*, 2012). Soil C:N of all the farms in this study were below 20-30 threshold, indicated the soils have higher

susceptibility to N mineralization that may lead to increased chances of N leaching, N<sub>2</sub>O and CO<sub>2</sub> emissions (Haney *et al.*, 2012; Huang *et al.*, 2004).

The SOM recorded at CF1 and OF2 were similar which was 3.7% and 3.6%, while CF2 and OF1 were 7.8% and 5.8%. It can be categorized based on the SOM percentages: very low (< 3%), low (3-7%), medium (7-17%), high (17-35%), very high (>35%) (Hill, 2015). Based on the classifications, the SOM in CF2 was in medium level category while the other farms were in low level. The increased of SOM is crucial in C sequestration as the C are mostly stored as organic matter in soil, study showed about 75% of soil C sequestration is from organic matter (Pacala *et al.*, 2001; Wofsy, 2001). In addition, SOM improves soil properties by increased soil water capacity, available water content in sandy soil and increases both air and water flow rates through fine textured soil (Mobar *et al.*, 2015).

The results indicated that the arable soil of OF1 contained highest amount of total potassium (K) which was around 1,135 ppm then followed by OF2 (875 ppm), CF2 (524 ppm), and CF1 (430 ppm). However, K concentrations in soil were lower as compared to the data recorded by Zhang *et al.* (2011) and Zörb *et al.* (2014) which the total K content in the top 20cm of most agricultural soils were between 10,000 ppm and 23,000 ppm. This might be due to the variation of soil types, fertilizer and crop types. For phosphorus (P), it is within the range of 1,000-2,000 ppm in the farms in this study which is similar to the soil properties reported by Zhang *et al.* (2011) where the soil P concentrations are between 450 ppm and 1,600 ppm. Manganese (Mn) was found to be the highest in OF1 soil (203 ppm), followed by CF1 (101 ppm), OF2 (67.8 ppm) and CF2 (62.8 ppm). Few authors reported the average total soil Mn was about 600 ppm and ranged between 20 ppm to 3000 ppm (Mousavi *et al.*, 2011; Schulte & Kelling, 1999).

The soils in the farms in this study contained 23.5, 18.9, 22.6, and 9.72 ppm of copper (Cu) in CF1, CF2, OF1, and OF2, respectively. The Cu concentrations were much lower as compared to the vineyard soil (65-87 ppm) but it is similar to the Cu content found in Aridisol (calcareous) soil which is about 24.5 ppm (Mackie *et al.*, 2013; Shaheen *et al.*, 2014). The soil Cu levels were classified based on low (< 0.25ppm), medium (0.25-0.5 ppm), high (>0.5 ppm) and the overall soil Cu content in the farms in this study fall under the high concentration category (Dinkins & Jones, 2013).

The calcium (Ca) concentrations in soils of CF1, OF1 and OF2 were higher than typical sandy soil test (400 to 500 ppm) but lower than clayey soils that usually contained Ca above 2,500 ppm (Espinoza *et al.*, 2006; Silva & Uchida, 2000; Yost & Uchida, 2000). The high Ca in soil of farms in this study suggests higher soil clay content or application of gypsum. The high Ca concentrations in OF2 might be due to the frequent applications of gypsum. According to farmers of OF1, the compost used in OF1 contained material such as chicken manure, fish mill, rice husks, and gypsums. Thus, the gypsum in composts has contributed to Ca input into soil. The CF1 is a mining reclamation area and the areas were filled with 50% clayey and 50% sandy soils. Thus, the high Ca concentration in CF1 is contributed by the high amount of clay in soil samples. The highest soil magnesium (Mg) concentration was from CF1 with 2,042 ppm, followed by OF1 with 1579 ppm, OF2 with 818 ppm and the lowest was CF2 about 374 ppm. Study indicated Mg in soil differed considerably which can range from 500 ppm to 5000 ppm due to high variations of Mg content in farm input materials (Gransee & Fühns, 2013). Research demonstrated that Ca and Mg levels in soils were highly correlated with the soil pH, in which lower soil acidity increased the Ca and Mg compound stability in soil (Upadhyay *et al.*, 2013). The pH results indicated CF1, OF1 and OF2 soils were neutral to moderately alkaline, while CF2 soil was slightly acidic to

neutral according to pH level classifications by Horneck et al. (2011). Thus, the slightly acidic soil in CF2 might lead to lower soil Ca and Mg due to lower compound stability as compared to other farms in this study.

The soil heavy metal pollution was evaluated by measuring the soil cadmium (Cd) and lead (Pb) concentrations. Cadmium is a contaminant that was often found in phosphorus fertilizer in a traceable amount, and it can build up in agriculture soil if the input via fertilizer is greater than the removal through crop harvest, erosion, leaching or bio-turbation (Grant, 2015). It was observed that no detectable Cd was found in the soils of the farms in this study. Lead is naturally present in soils at a range of 15 to 40 ppm and with a reading of more than 1000 ppm of Pb indicated soil pollution (Allen *et al.*, 2012). The results showed only OF1 soil contained detectable Pb and the concentration does not indicate heavy metal pollution.

#### **4.2 Material flow analysis (STAN model)**

The MFA models generated served as a structural layer for element flow in later part of the study (Section 4.3 and 4.4). All the relevant processes and materials were monitored and quantified during the study period and later modeled with STAN 2.5 software. The system boundary in this study was within the farm system of each farms in this study (Figure 4.1). The modeling only considered material that enters or leaves the farm systems (input-output). The key processes identified were “Farm land”, “Postharvest”, “Reuse”, and “Storage”. The “Farm land” is the vegetable cultivation process while “Postharvest” involved postharvest processes where the harvested vegetables were weighed, trimmed and/or washed.

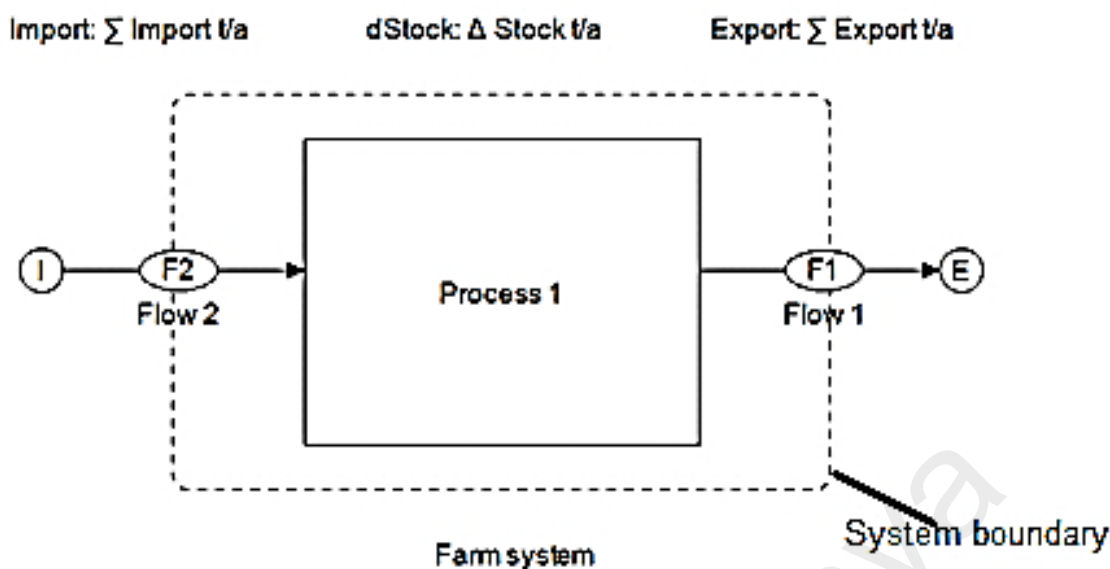


Figure 4.1: Schematic of system boundary for individual farm

The postharvest activities vary between farm systems due to differences in management styles. At conventional farm, the harvested vegetables were sold directly to wholesaler (CF1) or supermarket and restaurant (CF2) thus vegetables were only washed and weighted before it was placed into transportation baskets. On the other hand, the postharvest activities in OF1 involved weighing, washing, trimming and packaging that generate waste materials such as wastewater, organic wastes and plastic packaging wastes. However, the postharvest stage in OF2 was operated differently whereby the harvested vegetables were weighed and exported out from the farm system. The produce of OF2 were mainly exported to Singapore thus require to comply with the country's requirements. Thus, the farms would outsource the postharvest treatment, processing and packaging to a qualified packing center. Because the packing center is not located within the farm system boundary, thus it is excluded from the MFA model.

Plastic packaging wastes were generated from “Farmland” and were reused for various purposes thus go through the process of “Reuse”. On the other hand, the “Storage” is

the process where organic wastes or scrap plastic wastes were removed from “Farmland”, “Postharvest” or “Reuse” and stored within farm system with no further utilization. “Import” is the total material input to the farm systems: compost, irrigation, rainfall, chicken manure, chemical fertilizer, pesticide, herbicide, seeds, plastic packaging, and washing water into the farm system. “Export” is the total material output from the farm system which includes evapotranspiration, runoff and leaching, wastewater and vegetables. “dStock” is the differences between import and export. Based on the process and flow identified from site survey and STAN model, the farm metabolism schemes were generated and expressed in equations Eq. 4.1-4.4.

$$\text{MFA}_{\text{CF1}} = (\text{C} + \text{CM} + \text{IW} + \text{R} + \text{CF} + \text{P} + \text{H} + \text{S} + \text{PP} + \text{W}) - (\text{HV} + \text{E} + \text{RL} + \text{WW}) \quad (\text{Eq. 4.1})$$

$$\text{MFA}_{\text{CF2}} = (\text{CM} + \text{IW} + \text{R} + \text{CF} + \text{P} + \text{H} + \text{S} + \text{PP} + \text{W}) - (\text{HV} + \text{E} + \text{RL} + \text{WW}) \quad (\text{Eq. 4.2})$$

$$\text{MFA}_{\text{OF1}} = (\text{C} + \text{BC} + \text{VC} + \text{IW} + \text{PR} + \text{S} + \text{PM} + \text{PP} + \text{W}) - (\text{HV} + \text{E} + \text{RL} + \text{WW} + \text{PP}) \quad (\text{Eq. 4.3})$$

$$\text{MFA}_{\text{OF2}} = (\text{C} + \text{G} + \text{PR} + \text{S} + \text{PM} + \text{IW} + \text{PP}) - (\text{HV} + \text{E} + \text{RL}) \quad (\text{Eq. 4.4})$$

where, MFA is material flow analysis of CF1, CF2, OF1 and OF2; C, is compost; BC, Bokashi compost; VC, is vermicompost; CM, is chicken manure; PM, is peat moss; G, is gypsum; IW, is irrigation water; R, is rainfall; CF, is chemical fertilizer; P, is pesticide; PR, is pest repellent; H, is herbicide; S, is seed; PP, is plastic packaging; W, is



water used for washing activities; HV, is harvested vegetable; E, is evapotranspiration; RL, is runoff and leaching; WW, is waste water generated from washing activities.

Based on the farm metabolism generated, the input-output tables were tabulated with the major input-output and quantified materials. Table 4.2 shows that the major material used in CF1 were compost, chicken manure, and chemical fertilizer while major material output were vegetable, runoff and leachate.

Table 4.2: Material input-output in CF1

<b>Input</b>	<b>Unit</b>	<b>Total Average (<math>\pm</math> SD) Material per Hectare <sup>a</sup></b>
Compost	kg ha <sup>-1</sup> y <sup>-1</sup>	7020
Chicken Manure	kg y <sup>-1</sup>	18720
Chemical Fertilizer	kg y <sup>-1</sup>	8421
Irrigation Water	m <sup>3</sup> y <sup>-1</sup>	5316(1620)
Rainfall	m <sup>3</sup> y <sup>-1</sup>	26195(3207)
Washing Water (Postharvest)	m <sup>3</sup> y <sup>-1</sup>	5.07(1)
Pesticides	m <sup>3</sup> y <sup>-1</sup>	4.072
Herbicide	m <sup>3</sup> y <sup>-1</sup>	0.679
Seeds	kg y <sup>-1</sup>	179(100)
<b>Output</b>		
Total Vegetable Production	kg y <sup>-1</sup>	107171(60085)
Amaranth	kg y <sup>-1</sup>	16690(9502)
Choy Sum	kg y <sup>-1</sup>	56116(31200)
Hong Kong Choy Sum	kg y <sup>-1</sup>	17091(9632)
Water Spinach	kg y <sup>-1</sup>	17274(9751)
Evapotranspiration	m <sup>3</sup> y <sup>-1</sup>	16775(253)
Plastic Waste (Farm land)	kg y <sup>-1</sup>	961.96(0)
Vegetable Wastes (Farm land)	kg y <sup>-1</sup>	221(0)
Vegetable Wastes (Postharvest)	kg y <sup>-1</sup>	19.2(0)
Runoff and leaching	m <sup>3</sup> y <sup>-1</sup>	14736(1688)
Wastes Water (Postharvest)	m <sup>3</sup> y <sup>-1</sup>	5.07(0)

<sup>a</sup> SD: standard deviation (value in bracket)

Table 4.3 demonstrates the major input and output material in CF2. Similar with CF1, the major material used were chicken manure and chemical fertilizer. However,

compost was not used in CF2. The major material outputs in CF2 were vegetable, runoff and leaching.

Table 4.3: Material input-output in CF2

<b>Input</b>	<b>Unit</b>	<b>Total Average (<math>\pm</math> SD) Material per Hectare <sup>a</sup></b>
Chicken Manure	kg y <sup>-1</sup>	57030
Chemical Fertilizer	kg y <sup>-1</sup>	5941
Irrigation Water (Farm Land)	m <sup>3</sup> y <sup>-1</sup>	9920(1542)
Rainfall	m <sup>3</sup> y <sup>-1</sup>	22780(2278)
Washing Water (Postharvest)	m <sup>3</sup> y <sup>-1</sup>	1.84(0)
Pesticides	m <sup>3</sup> y <sup>-1</sup>	2.31
Herbicide	m <sup>3</sup> y <sup>-1</sup>	0.58
Seeds	kg y <sup>-1</sup>	17.82(1.980)
<b>Output</b>		
Total Vegetable Production	kg y <sup>-1</sup>	10693(1190)
Amaranth	kg y <sup>-1</sup>	1782.2(198.4)
Choy Sum	kg y <sup>-1</sup>	1782.2(198.4)
Japanese Choy Sum	kg y <sup>-1</sup>	1782.2(198.4)
Spring Onion	kg y <sup>-1</sup>	1782.2(198.4)
Water Spinach	kg y <sup>-1</sup>	1782.2(198.4)
Xiao Pak Choy	kg y <sup>-1</sup>	1782.2(198.4)
Evapotranspiration	m <sup>3</sup> y <sup>-1</sup>	16775(253)
Plastic Waste (Farm land)	kg y <sup>-1</sup>	374.3(0)
Vegetable Wastes (Postharvest)	kg y <sup>-1</sup>	21.7(0)
Runoff and leaching	m <sup>3</sup> y <sup>-1</sup>	15925(1517)
Waste Water (Postharvest)	m <sup>3</sup> y <sup>-1</sup>	1.84(0)

<sup>a</sup> SD: standard deviation (value in bracket)

Table 4.4 shows that the major input in OF1 was compost and three types of compost were used: normal compost, Bokashi compost and vermicompost. Each type of compost was applied at different planting stage. Normal compost was applied as basal fertilizer before planting, vermicompost was used as potting material for seeding, and Bokashi compost was applied one week after seedling transplant from nursery to soil bed. Vegetable production was major material output which consists of 17 types of crop variety. Besides vegetable production, the output of runoff and leaching was also one of the key materials in OF1.

Table 4.4: Material input-output in OF1

<b>Input</b>	<b>Unit</b>	<b>Total Average (<math>\pm</math> SD) Material per Hectare <sup>a</sup></b>
Normal Compost	kg y <sup>-1</sup>	187500
Bokashi Compost	kg y <sup>-1</sup>	18750
Vermicompost	kg y <sup>-1</sup>	600
Peat Moss	kg y <sup>-1</sup>	2880(1222)
Irrigation Water (Farm Land)	m <sup>3</sup> y <sup>-1</sup>	19283(3148)
Irrigation Water (Nursery)	m <sup>3</sup> y <sup>-1</sup>	4861(486.1)
Washing Water (Postharvest)	m <sup>3</sup> y <sup>-1</sup>	0.115(0)
Pest Repellent	m <sup>3</sup> y <sup>-1</sup>	5.00
Seeds	kg y <sup>-1</sup>	57.50(24.50)
<b>Output</b>		
Total Vegetable Production	kg y <sup>-1</sup>	34561(14665)
Amananth	kg y <sup>-1</sup>	1808(603)
Amananth, Red	kg y <sup>-1</sup>	722(717)
Fu Gui Choy	kg y <sup>-1</sup>	770(585)
Fu Mak	kg y <sup>-1</sup>	650(221)
Hong Kong Choy Sum	kg y <sup>-1</sup>	4193(1140)
Japanese Choy Sum	kg y <sup>-1</sup>	976(786)
Kai Lan	kg y <sup>-1</sup>	3529(1014)
Lettuce	kg y <sup>-1</sup>	2761(1172)
Mini Cos Lettuce	kg y <sup>-1</sup>	915(456)
Nai Bai	kg y <sup>-1</sup>	3561(1547)
New Zealand Spinach	kg y <sup>-1</sup>	92(86)
Ong King Pak Choy	kg y <sup>-1</sup>	3142(1446)
Senposai	kg y <sup>-1</sup>	3091(1118)
Sweet Potato Leaf	kg y <sup>-1</sup>	189(418)
Water Spinach	kg y <sup>-1</sup>	1817(735)
Xiao Pak Choy	kg y <sup>-1</sup>	3308(1311)
Xiu Zhen Choy Sum	kg y <sup>-1</sup>	3037(1311)
Evapotranspiration	m <sup>3</sup> y <sup>-1</sup>	8562(129)
Rejected Vegetables	kg y <sup>-1</sup>	1697(0)
Plastic Waste (Postharvest)	kg y <sup>-1</sup>	246.38(0)
Plastic Waste (Farm Land)	kg y <sup>-1</sup>	394.5(0)
Runoff and leaching	m <sup>3</sup> y <sup>-1</sup>	10721(3019)
Wastes Water (Postharvest)	m <sup>3</sup> y <sup>-1</sup>	0.115(0)

<sup>a</sup> SD: standard deviation (value in bracket)

Table 4.5 shows the key material input and output at OF2. Similar to OF1 compost is the major input in OF2 however, the amount used was six times lower than OF1. In

addition, only one type of compost used in OF2 which was NASAA certified organic compost. Vegetable production was the main material output which consists of 9 types of vegetable varieties.

Table 4.5: Material input-output at OF2

<b>Input</b>	<b>Unit</b>	<b>Total Average (<math>\pm</math> SD) Material per Hectare <sup>a</sup></b>
Compost (Midori 333)	kg y <sup>-1</sup>	34180
Gypsum	kg y <sup>-1</sup>	5340
Peat Moss	kg y <sup>-1</sup>	9516(7498)
Irrigation Water (Farm Land)	m <sup>3</sup> y <sup>-1</sup>	15815(7260)
Irrigation Water (Nursery)	m <sup>3</sup> y <sup>-1</sup>	5509(551)
Pest Repellent	m <sup>3</sup> y <sup>-1</sup>	1.678
Seeds	kg y <sup>-1</sup>	190(150)
<b>Output</b>		
Total Vegetable Production	kg y <sup>-1</sup>	114191(89976)
Vegetable		
Amananth	kg y <sup>-1</sup>	1209(1361)
Amananth, Red	kg y <sup>-1</sup>	452(675)
Choy Sum	kg y <sup>-1</sup>	60341(60632)
Lettuce	kg y <sup>-1</sup>	432(1137)
Nai Bai	kg y <sup>-1</sup>	1009(1312)
New Zealand Spinach	kg y <sup>-1</sup>	18634(4803)
Sweet Potato Leaf	kg y <sup>-1</sup>	5025(3176)
Water Spinach	kg y <sup>-1</sup>	22584(10841)
Xiao Pak Choy	kg y <sup>-1</sup>	4505(6039)
Evapotranspiration	m <sup>3</sup> y <sup>-1</sup>	8562(129)
Plastic Waste (Farm land)	kg y <sup>-1</sup>	3965(0)
Runoff and leaching	m <sup>3</sup> y <sup>-1</sup>	7253(7131)

<sup>a</sup> SD: standard deviation (value in bracket)

The quantified input-output materials for each farm were incorporated into the graphical model by the use of the STAN software. The STAN graphical models provide visual images of the farm's material flow and the software reconciled the data by altering the mean values of uncertain data to remove contradictions of data. The major material input observed in all the farms were water and biomass input (Figure 4.2). Water is a crucial element in crop growth and therefore irrigation water is the important input in

the farm system (Birkenshaw & Bailey, 2003). The volume of irrigation water is highly dependent on the cultivation methods, crop types, soil water holding capacity, irrigation system, landscape, climate, and farm management (Birkenshaw & Bailey, 2003; Connellan, 2002)

Each farmer has their own sets of irrigation management. However, all four farms in this study were established with similar overhead sprinkler irrigation system. Yearly, the volume of irrigation water in CF1, CF2, OF1, and OF2 was  $5,316 \pm 1,620$ ,  $9,920 \pm 1,512$ ,  $19,282 \pm 3,148$ , and  $15,815 \pm 7,260 \text{ t ha}^{-1} \text{ y}^{-1}$  and similar irrigation are repeated by Song et al. (2009). However, the irrigation rates in this study were lower than the rates reported by Abdulkadir et al. (2013) who estimated the irrigation rates of  $14,277 \pm 11,719$  and  $21,617 \pm 6,464 \text{ t y}^{-1} \text{ ha}^{-1}$  in peri-urban and urban commercial gardening.

Generally, the water management levels of the farms in this study were classified as level two water management with systematic irrigation (Simonne *et al.*, 2004). The higher irrigation volumes in organic systems (OF1 and OF2) were due to the rain shelter that prevents rainfall from entering the “Farm land” directly. In contrast, the open field planting in conventional system (CF1 and CF2) allows rainfall to supplement the sprinkler irrigation and lower volume of irrigation is required to meet the crop demand. According to growers of CF1 and CF2, the irrigation system was not used during raining days. The average rainfall input recorded for CF1 and CF2 was about  $26,194 \pm 3,207$  and  $22,779 \pm 2,278 \text{ t}^{-1} \text{ ha}^{-1} \text{ y}^{-1}$  (Figure 4.2 and 4.3). This makes the total water input through irrigation and rainfall in CF1 and CF2 was around  $31,510$  and  $32,699 \text{ t}^{-1} \text{ ha}^{-1} \text{ y}^{-1}$  which was higher as compared to the organic system OF1 and OF2.

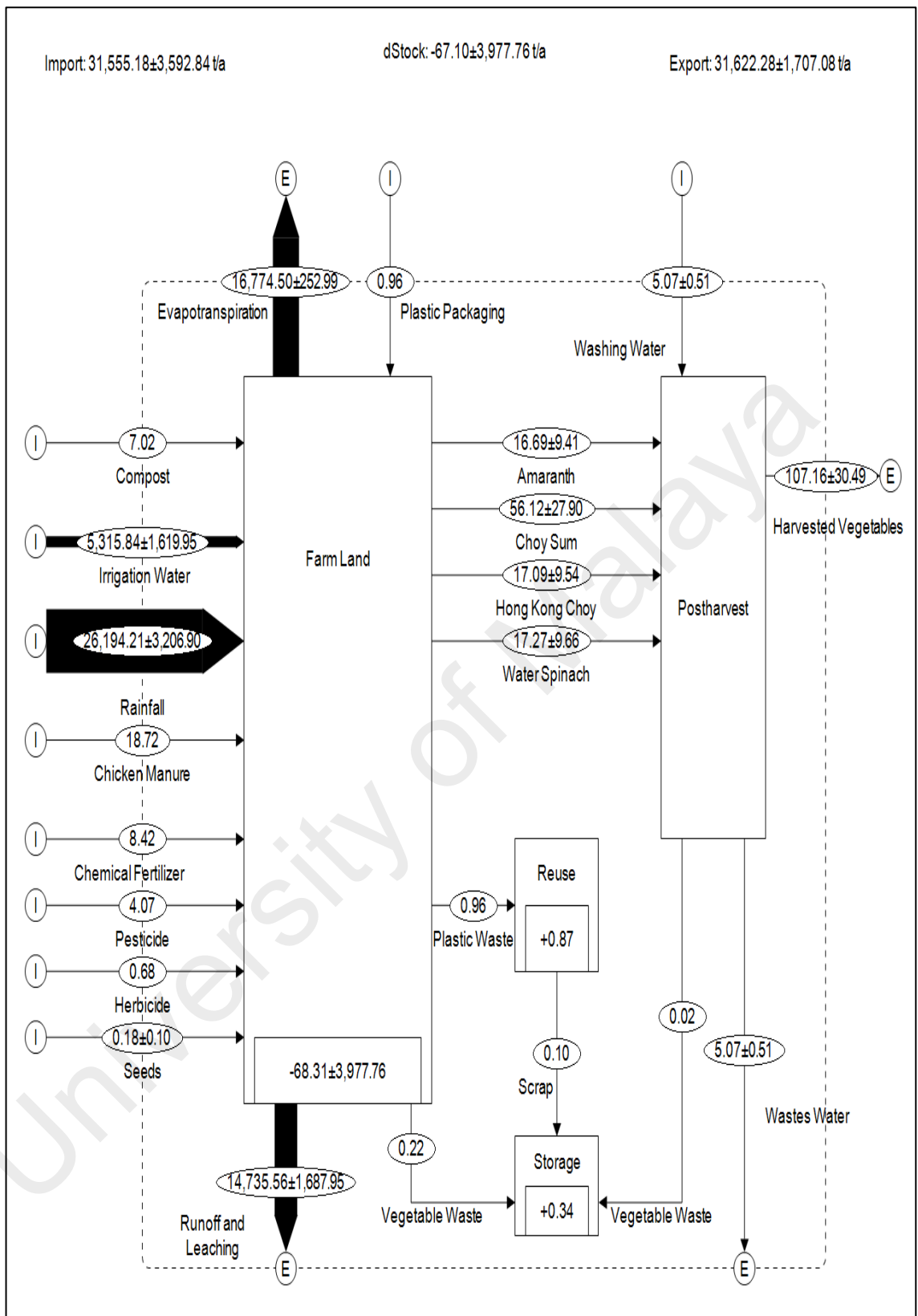


Figure 4.2: Material flow of conventional farm CF1 ( $t^{-1} ha^{-1} y^{-1}$ ).

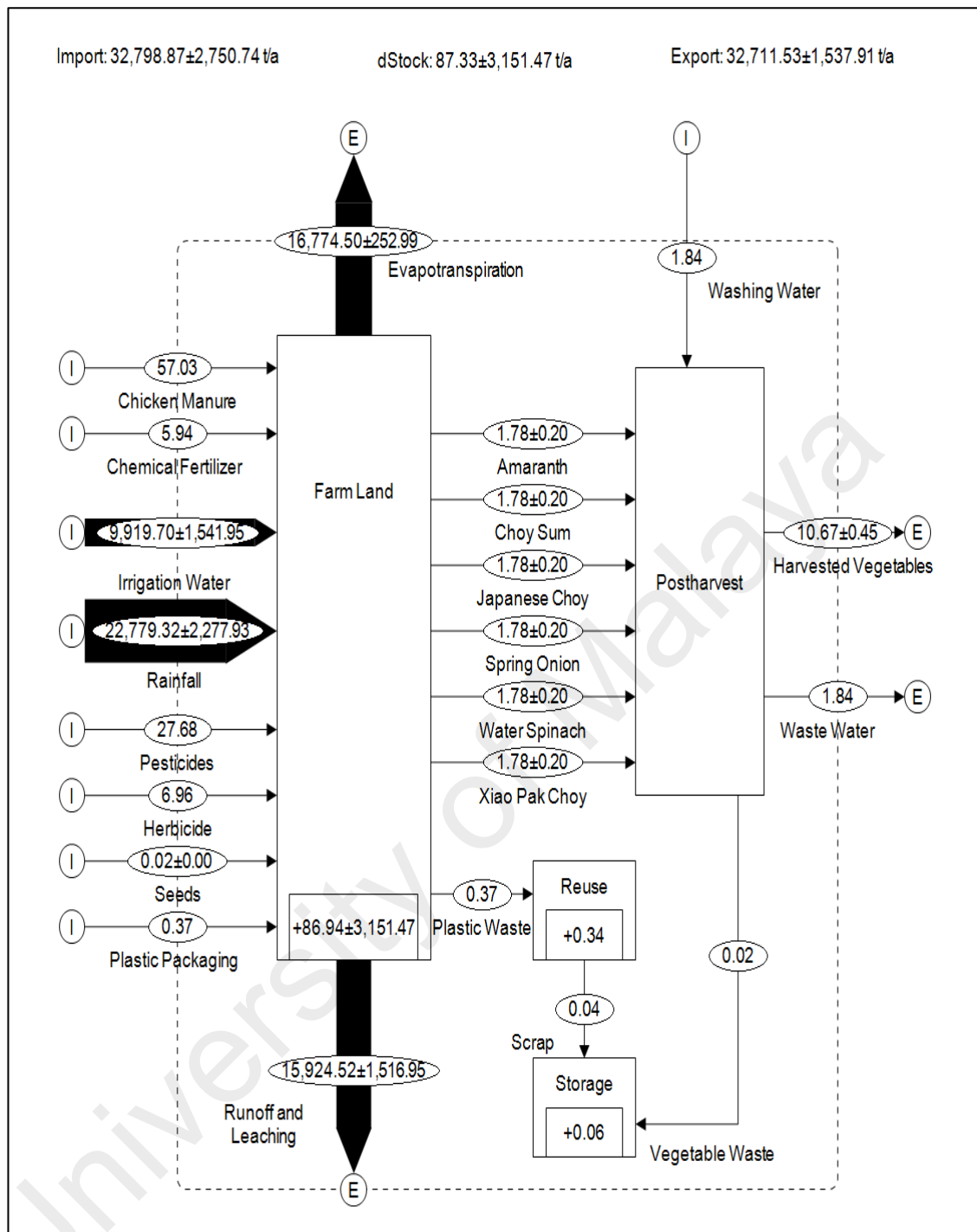


Figure 4.3: Material flow of conventional farm CF2 ( $t^{-1} ha^{-1} y^{-1}$ ).

Chemical fertilizer, chicken manure, and compost were the typical nutrient inputs in the farms. Chemical fertilizer used were only observed in conventional systems in which around 8.42 in CF1 and  $5.94 t^{-1} ha^{-1} y^{-1}$  in CF2 were used yearly. Chicken manure were also used in CF1 and CF2 as soil amendments to promote crop growth and it is a common practice in local farms (Amos *et al.*, 2013; Cheung & Wong, 1983; Tiraieyari

*et al.*, 2014). The average application of chicken manure was about 18.72 and 57.03 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> in CF1 and CF2, respectively. A survey of 200 vegetables farms in China showed the average manure application rates ranged between 36.5 and 54.0 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> during year 1996 to 2000 (Chen *et al.*, 2004). The manure application rate in CF1 was lower than the data reported by Chen *et al.* (2004) while CF2 was slightly higher.

In this study, only conventional farms applied chicken manure as soil amendments. This is because the certified organic farms were restricted or discouraged from using fresh or untreated manure with the objective to minimize pathogenic contamination on produce (USDA, 2011; Winter & Davis, 2006). Thus, compost was the only nutrient source in organic systems. At OF1, three types of composts were applied at different stages of planting cycle. Normal composts (187.5 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup>) were added to the soil as base fertilizer during soil tillage (Figure 4.4). After one week of seedling transplant from nursery to soil bed, about 18.75 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> of Bokashi compost (compost with effective microbes) were applied. Vermicompost was another type of compost used in OF1 as potting materials in the nursery to promote seedling growth by supplying nutrient for crop development (Clark & Cavigelli, 2005; Kumar & Raheman, 2012; Raviv *et al.*, 1998). Annually, about 0.23 ± 0.20 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> of vermicompost were mixed with peat moss (1:1 ratio) to make up potting materials for vegetable seeds growth in nurseries. In contrast, OF2 used only peat moss as potting materials with an estimated average of 7.08 ± 5.58 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup>.

On the other hand, the yearly compost application in OF2 was 34.18 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> (Figure 4.5). According to Evanylo *et al.* (2008), in order to achieve agronomic benefit, the compost application rate should be 20% of the crop N requirement which is about 32.9 to 66.9 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> of compost (Evanylo *et al.*, 2008). The compost application rate at



OF2 is within the range proposed by Evanylo et al. (2008). However, at OF1 the compost rate is almost triple the recommended value.

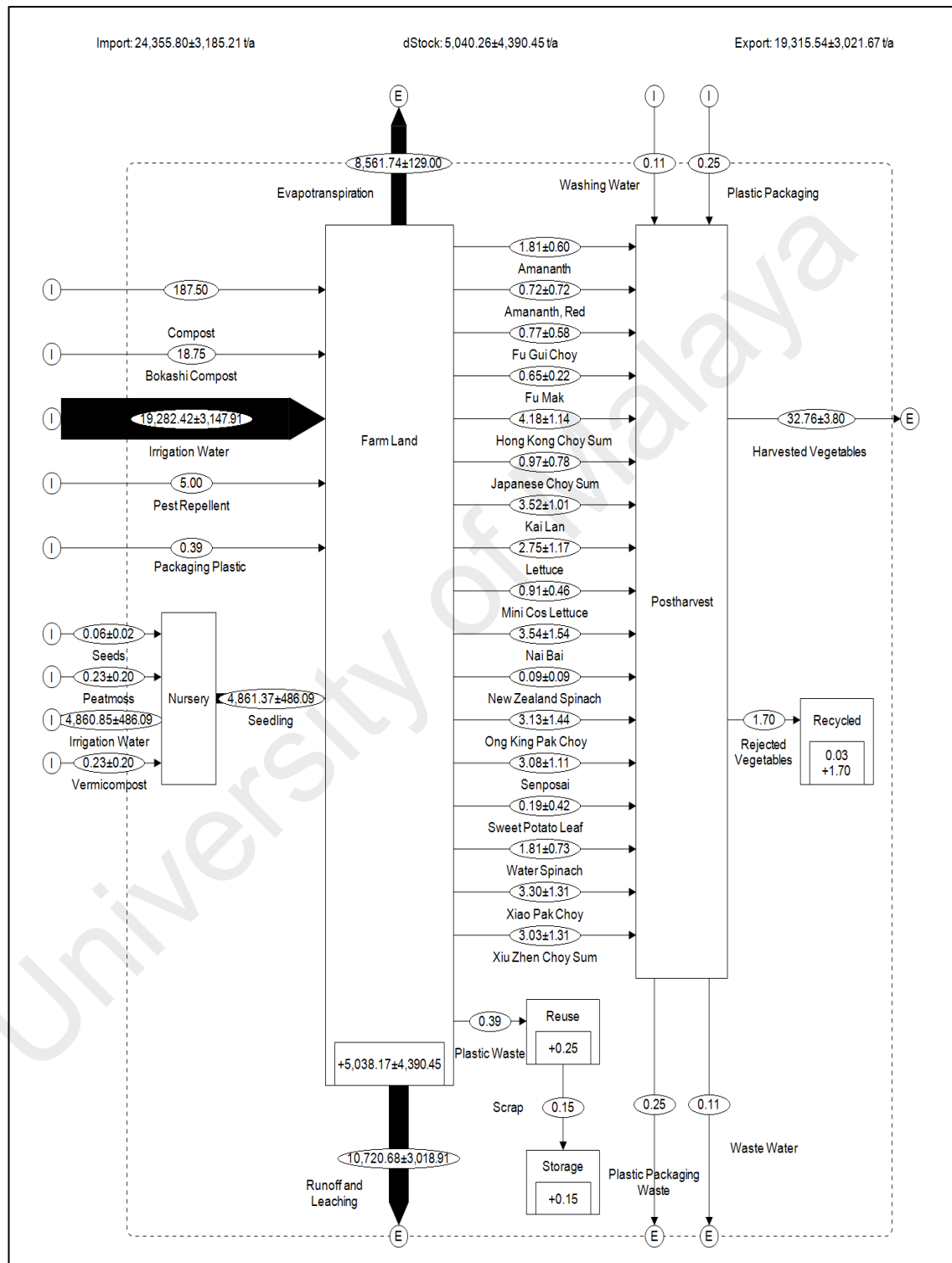


Figure 4.4: Material flow of organic farm OF1 ( $t^{-1} ha^{-1} y^{-1}$ ).

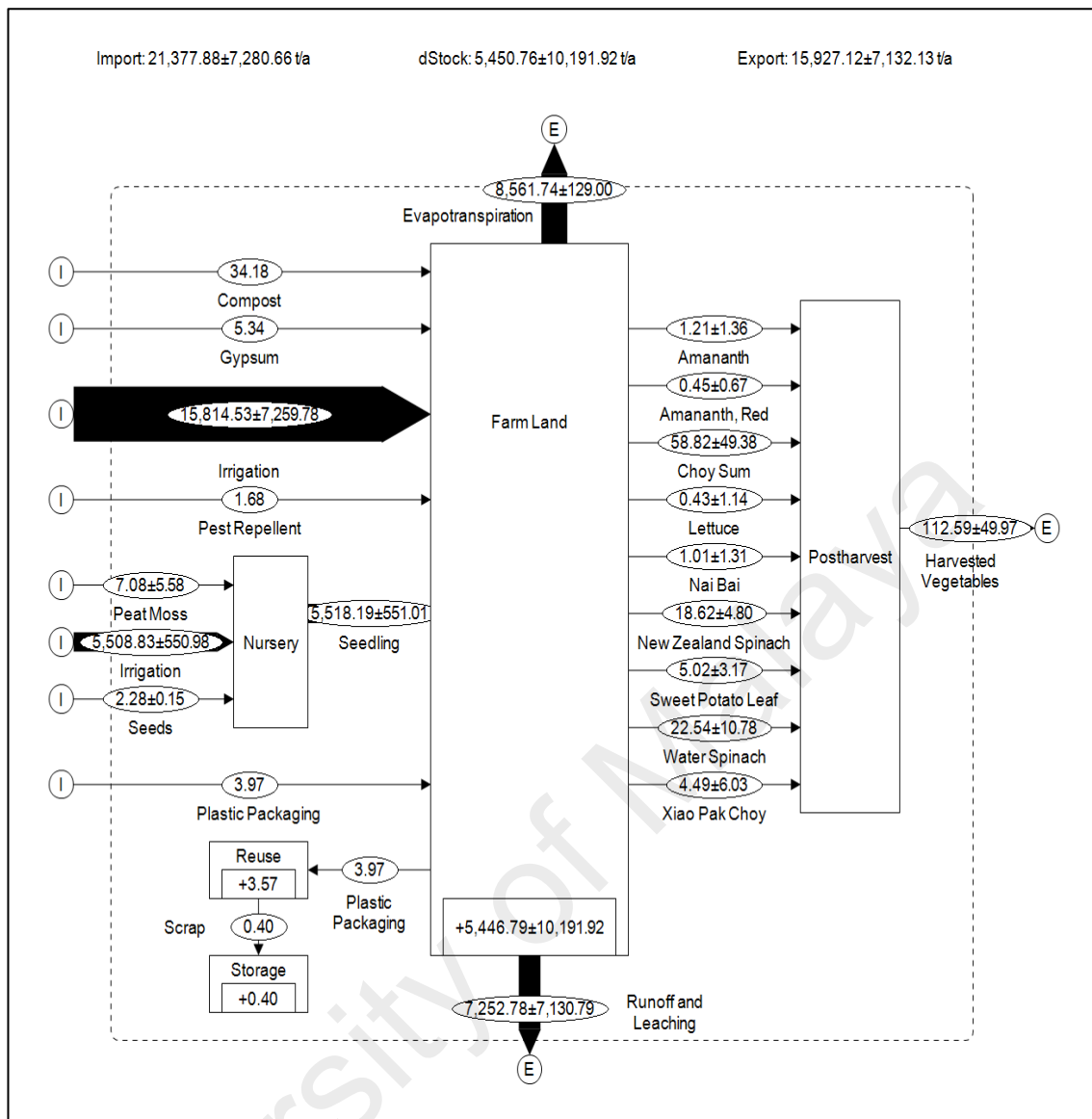


Figure 4.5: Material flow of organic farm OF2 ( $t^{-1} ha^{-1} y^{-1}$ ).

Pesticide and weed control at conventional systems relied on pesticide and herbicide. The pesticide and herbicide flow in the MFA is the amount sprayed onto field after water dilution. The total amount of pesticide sprayed in CF1 and CF2 was 4.07 and 27.68  $t^{-1} ha^{-1} y^{-1}$  whereas the herbicide was 0.68 and 6.92  $t^{-1} ha^{-1} y^{-1}$ , respectively. Pesticide was applied every 5-7 days in conventional systems dependent on the weather and the severity of the pest problems. Herbicide was usually applied after crop harvesting. Because of the chemical use restrictions in organic systems, OF1 and OF2 relied on organic certified pest repellent to control pest and mechanical weed control.

The annual diluted pest repellent volume recorded was about 5.0 and 1.68 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> in OF1 and OF2 with application frequency of every 7-10 days. The types of pesticide, herbicide and pest repellent used were not disclosed by the farmers. The application frequencies were similar with the data reported by a study in Cameron Highland whereby majority of the farmers applied pesticide every 7-9 days during dry season and 4-6 days during wet season (Mazlan & Mumford, 2005).

In general, the main outputs of the farm systems were harvested vegetable, runoff and leaching. Harvested vegetable is the crop production from the farm systems and it is also the main biomass output. Among the four farms in this study OF2 and CF1 recorded similar yield which was about 113 ± 50 and 107 ± 30 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup>. It is followed by OF2 (33 ± 3.8 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup>) and CF2 with 11 ± 0.45 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> of vegetables yield. The results were differed from de Ponti et al. (2012) where the organic yield was reported to be 80 ± 21% of conventional production. Study suggested that good management practices coupled with appropriate crop types and growing conditions will allow the organic systems to match conventional yields (Seufert *et al.*, 2012). The runoff and leaching were calculated based on the water model which estimated potential volume of runoff and leaching leaving the farm system (Bengtsson *et al.*, 2003). It is estimated there was 14,736 ± 1,688, 15,925 ± 1,517, 10,721 ± 3,019, and 7,253 ± 7,131 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> of potential water output from CF1, CF2, OF1 and OF2, respectively via runoff and leaching. Approximately 87.33 ± 3,152, 5,043 ± 4,390 and 5,453 ± 10,191 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup> of material stock (dStock) in CF1, OF1 and OF2, respectively, which indicates higher input of material as compared to the output. However, the annum stock change in CF2 (-67 ± 3,978 t<sup>-1</sup> ha<sup>-1</sup> y<sup>-1</sup>) showed higher output as compared to the input of the system. The high uncertainty values of dStock value signified high variation of material flow in the farm systems. Three stock processes were identified in CF1, CF2 and OF2 while

four stock processes were observed at OF1. The stock processes are the processes where the input materials were deposited as stock. Majority of the materials were deposited into “Farm Land” as stock. About  $-68 \pm 3,978$ ,  $+87 \pm 3,152$ ,  $+5,041 \pm 4,391$ , and  $+5,449 \pm 10,192 \text{ t}^{-1} \text{ ha}^{-1} \text{ y}^{-1}$  of stock change in “Farm Land” process of CF1, CF2, OF1 and OF2, respectively. Other stock processes such as “Reuse”, “Storage” and “Recycled” only contained small amount of material deposition.

Overall, the farm systems vary in the amount of fertilizer, compost, manure, peat moss, and seeds input. The differences in water management and irrigation systems lead to variation in water inflow and outflow volume. Other than that, organic and conventional farm systems have different nutrient sources, pest and weed control regime that also lead to variation in the material flow. Other study also point out the organic and conventional systems differed in the amount of irrigation volume, rainfed volume, fertilizers amounts, and organic matter applications (Mitchell *et al.*, 2007).

#### **4.2.1 Agriculture wastes**

The proportion of wastes output from CF1, CF2, OF1 and OF2 were generally low. Plastic waste was the main waste type generated during the study period. Similarly, study by Goenster *et al.* (2014) also indicated plastic wastes as the major wastes produced from vegetable farm which is about 50% of total weight of wastes. The plastic wastes in this study were from packaging of consumable items such as fertilizer, compost, gypsum, pesticide, herbicide and pest repellent. Each year, about 0.96, 0.37, 0.39 and  $3.96 \text{ t ha}^{-1}$  of packaging wastes were generated from CF1, CF2, OF1 and OF2, respectively. The plastic wastes generated were stored within the farm and reuse for various purposes. There is lack of evidence for proper disposal, efficient waste management and recycling activities in the farms in this study. The plastic wastes

generated from farm systems can be a potential source for plastic recycling (Miskolczi *et al.*, 2009).

Organic waste were often assumed to be produced from “Farm land”. However, this study revealed that there was a small quantity of organic waste generated from “Postharvest” in CF1, CF2 and OF1. The trimming process at “Postharvest” stage is the main activity that produced organic waste. Organic waste was not generated in “Postharvest” stage of OF2 because this stage only involved weighing and chilling process. The organic wastes generated from “Postharvest” stage were usually left on the non-arable land without any treatment. However, in OF1 the organic wastes were recycled as animal feed. During the study period, there was around  $0.22 \text{ t ha}^{-1} \text{ year}^{-1}$  organic residues removed from “Farm land” in CF1. However, the usual practice in most farms is leaving organic residues on arable land to replenish soil nutrient (Palm *et al.*, 1997). According to the owner of CF1, the organic residues were removed from arable land as a quarantine measure to prevent the pest from spreading to other unaffected plot.

The C and N input of plastic packaging were low which resulted in C flux of 11.93, 4.64, 4.90, and 49.2  $\text{kg C ha}^{-1} \text{ y}^{-1}$  and N flux of 0.96, 0.37, 0.39, and 3.97  $\text{kg N ha}^{-1} \text{ y}^{-1}$  in CF1, CF2, OF1, and OF2, respectively. Possible reason for the low C and N input may because of the low C and N content which were about 1.24% and 0.1%. In addition, the low volume of plastic wastes also one of the factor that resulted the low C and N input in the elemental flows. The C and N input via plastic packaging were stocked or scrapped under “Reuse” or “Storage” in the farm system and do not contribute to soil C and N stock in “Farmland”.

### 4.3 C flow analysis (STAN model)

The material flows generated from the previous section (Section 4.2) provides a structure layer for C and N flow analysis in this section. Main objective of C flow was to evaluate potential C stock that can be stored in farm systems for C sequestration. Also, to identify potential C sequesters. Based on the farm survey and STAN model, the C flux metabolisms for the farms in this study were expressed in equations as shown in Eq. 4.5, 4.6, 4.7, and 4.8. The C flux metabolism included material that contributed to C mass input-output in the farm system.

$$CF_{CF1} = (C_{cm} + CM_{cm} + IW_{cm} + R_{cm} + CF_{cm} + TCP + S_{cm} + PP_{cm} + W_{cm}) - (HV_{cm} + G_{cm} + RL_{cm} + WW_{cm}) \quad (\text{Eq. 4.5})$$

$$CF_{CF2} = (CM_{cm} + IW_{cm} + R_{cm} + CF_{cm} + TCP + S_{cm} + PP_{cm} + W_{cm}) - (HV_{cm} + G_{cm} + RL_{cm} + WW_{cm}) \quad (\text{Eq. 4.6})$$

$$CF_{OF1} = (C_{cm} + BC_{cm} + VC_{cm} + IW_{cm} + TCP + S_{cm} + PM_{cm} + PP_{cm} + W_{cm}) - (HV_{cm} + G_{cm} + RL_{cm} + WW_{cm} + PW_{cm}) \quad (\text{Eq. 4.7})$$

$$CF_{OF2} = (C_{cm} + TCP + S_{cm} + PM_{cm} + IW_{cm} + PP_{cm}) - (HV_{cm} + G_{cm} + RL_{cm}) \quad (\text{Eq. 4.8})$$

where, CF is C flux of CF1, CF2, OF1 and OF2 (positive value indicated potential C stock while negative value indicated potential C lost from the system);  $C_{cm}$ , is C mass of compost;  $BC_{cm}$ , is C mass of Bokashi compost;  $VC_{cm}$ , is C mass of vermicompost;  $CM_{cm}$ , is C mass of chicken manure; TCP, is the total C derived from photosynthesis;  $PM_{cm}$ , is C mass of peat moss;  $IW_{cm}$ , is C mass of irrigation water;  $R_{cm}$ , is C mass of rainfall;  $S_{cm}$ , is C mass of seed;  $PP_{cm}$ , is C mass of plastic

packaging; W, cm, is C mass of washing water; HV, cm, is C mass of harvested vegetable; G, cm, is C mass of gaseous emissions (C dioxide, C monoxide, and methane); RL, cm, is C mass of runoff and leaching; WW, cm, is C mass of waste water.

Table 4.6 shows the estimated input-output of C mass and the C concentrations for the various commodities in CF1. The high vegetable production has resulted in considerable amount of C input in CF1.

Table 4.6: C input-output in CF1

<b>Input</b>	<b>Unit</b>	<b>Average C Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) C per Hectare <sup>c</sup></b>
Compost	kg y <sup>-1</sup>	36.10	1,794
Chicken Manure	kg y <sup>-1</sup>	20.44	2,686
Chemical Fertilizer	kg y <sup>-1</sup>	38.50	3,210
Irrigation Water	kg y <sup>-1</sup>	0.579 <sup>a</sup>	3.1(0.47)
Rainfall	kg y <sup>-1</sup>	0.222 <sup>a</sup>	5.8(0.51)
Washing Water (Postharvest)	kg y <sup>-1</sup>	0.579 <sup>a</sup>	0.0029(0.0001)
Seeds	kg y <sup>-1</sup>	30	37.5(21.0)
Total C derived from photosynthesis	kg y <sup>-1</sup>	36.2	3803(2132)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	36.2	2,717(1,523)
Amaranth	kg y <sup>-1</sup>	34.8	465(265)
Choy Sum	kg y <sup>-1</sup>	37.7	1,903(1,058)
Hong Kong Choy Sum	kg y <sup>-1</sup>	35.7	549(309)
Water Spinach	kg y <sup>-1</sup>	36.7	507(286)
C Monoxide	g min <sup>-1</sup>	NA	0.0888(0.0336)
C Dioxide	g min <sup>-1</sup>	NA	3.48(1.89)
Plastic Waste (Farm Land)	kg y <sup>-1</sup>	1.24	11.93
Vegetable Wastes (Farm Land)	kg y <sup>-1</sup>	36.2	20.01
Vegetable Wastes (Weighing Station)	kg y <sup>-1</sup>	36.2	1.739
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	198(7.5)
Wastes Water (Postharvest)	kg y <sup>-1</sup>	0.579 <sup>a</sup>	0.0071

<sup>a</sup> C concentration in mg per litre (mg L<sup>-1</sup>).

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. C concentration of runoff is 56.7 mg L<sup>-1</sup> and leaching is 141.2 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket).

NA: Not available

Table 4.7 shows the major C input and output in CF2. The main biomass C input was from chicken manure while main biomass C output was from vegetable production. Runoff and leaching was also one of the key output of C in CF2.

Table 4.7: C input-output in CF2

<b>Input</b>	<b>Unit</b>	<b>Average C Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) C per Hectare<sup>c</sup></b>
Chicken Manure	kg y <sup>-1</sup>	24.93	9,980
Chemical Fertilizer	kg y <sup>-1</sup>	38.5	2,264
Irrigation Water (Farm Land)	kg y <sup>-1</sup>	9.91 <sup>a</sup>	98(12)
Rainfall	kg y <sup>-1</sup>	1.59 <sup>a</sup>	36.1(0.8)
Washing Water (Postharvest)	kg y <sup>-1</sup>	9.91 <sup>a</sup>	0.018(0.0)
Seeds	kg y <sup>-1</sup>	30	3.7(0.4)
Total C derived from photosynthesis	kg y <sup>-1</sup>	NA	358(40)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	34.14	255.6(28.5)
Amaranth	kg y <sup>-1</sup>	33.90	48.3(5.38)
Choy Sum	kg y <sup>-1</sup>	31.77	51.0(5.67)
Japanese Choy Sum	kg y <sup>-1</sup>	34.67	49.4(5.50)
Spring Onion	kg y <sup>-1</sup>	36.88	46.0(5.12)
Water Spinach	kg y <sup>-1</sup>	35.20	50.2(5.59)
Xiao Pak Choy	kg y <sup>-1</sup>	32.42	46.2(5.15)
C Monoxide	g min <sup>-1</sup>	NA	0.006(0.006)
C Dioxide	g min <sup>-1</sup>	NA	4.09(3.45)
Plastic Waste (Farm land)	kg y <sup>-1</sup>	1.24	4.6408
Vegetable Wastes (Postharvest)	kg y <sup>-1</sup>	34.1	0.5182
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	392.6(0.96)
Waste Water (Postharvest)	kg y <sup>-1</sup>	9.91 <sup>a</sup>	0.0477(0.00)

<sup>a</sup> C concentration in mg per litre (mg L<sup>-1</sup>).

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. C concentration of runoff is 14.47 mg L<sup>-1</sup> and leaching is 20.56 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket).

NA: Not available

Table 4.8 shows the major C input output in OF1. The key C output was vegetable production and runoff and leaching. The high compost application in OF1 has resulted in high amount of C input.



Table 4.8: C input-output in OF1

<b>Input</b>	<b>Unit</b>	<b>Average C Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) C per Hectare <sup>c</sup></b>
Normal Compost	kg y <sup>-1</sup>	11.79	15,655
Bokashi Compost	kg y <sup>-1</sup>	22.42	2,976
Vermicompost	kg y <sup>-1</sup>	12.032	51
Peat Moss	kg y <sup>-1</sup>	17.345	375(159)
Irrigation Water (Farm Land)	kg y <sup>-1</sup>	3.47 <sup>a</sup>	67(4)
Irrigation Water (Nursery)	kg y <sup>-1</sup>	3.47 <sup>a</sup>	17(1)
Washing Water (Postharvest)	kg y <sup>-1</sup>	3.47 <sup>a</sup>	0.0004(0.00)
Seeds	kg y <sup>-1</sup>	30	12.1(5.1)
Total C derived from photosynthesis	kg y <sup>-1</sup>	NA	1,193(506)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	35.2	852(362)
Amananth	kg y <sup>-1</sup>	33.54	49(16)
Amananth, Red	kg y <sup>-1</sup>	34.95	20(20)
Fu Gui Choy	kg y <sup>-1</sup>	31.7	12(9)
Fu Mak	kg y <sup>-1</sup>	39.9	21(7)
Hong Kong Choy Sum	kg y <sup>-1</sup>	30.9	117(32)
Japanese Choy Sum	kg y <sup>-1</sup>	35.7	28(22)
Kai Lan	kg y <sup>-1</sup>	33.8	119(34)
Lettuce	kg y <sup>-1</sup>	35.9	40(17)
Mini Cos Lettuce	kg y <sup>-1</sup>	34.5	13(6)
Nai Bai	kg y <sup>-1</sup>	34.1	61(26)
New Zealand Spinach	kg y <sup>-1</sup>	38.7	3(3)
Ong King Pak Choy	kg y <sup>-1</sup>	34.6	65(30)
Senposai	kg y <sup>-1</sup>	35.0	43(16)
Sweet Potato Leaf	kg y <sup>-1</sup>	41.1	7(15)
Water Spinach	kg y <sup>-1</sup>	35.0	51(21)
Xiao Pak Choy	kg y <sup>-1</sup>	36.3	96(38)
Xiu Zhen Choy Sum	kg y <sup>-1</sup>	33.2	91(39)
C Monoxide	g min <sup>-1</sup>	NA	0.1262(0.0432)
C Dioxide	g min <sup>-1</sup>	NA	36.0(24.7)
Rejected Vegetables	kg y <sup>-1</sup>	35.2	41.8
Plastic Waste (Postharvest)	kg y <sup>-1</sup>	1.24	3.1
Plastic Waste (Farm Land)	kg y <sup>-1</sup>	1.24	4.9
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	658.5(12.15)
Wastes Water (Postharvest)	kg y <sup>-1</sup>	3.47 <sup>a</sup>	0.0021

<sup>a</sup> C concentration in mg per litre (mg L<sup>-1</sup>)

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. C concentration of runoff is 29.11 mg L<sup>-1</sup> and leaching is 65.12 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket)

NA: Not available

Table 4.9 shows the C input and output in OF2. Similarly, photosynthesis is an important C input. However, compost contributed highest amount of C into OF2.

Table 4.9: C input-output in OF2

<b>Input</b>	<b>Unit</b>	<b>Average C Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) C per Hectare <sup>c</sup></b>
Compost (Midori 333)	kg y <sup>-1</sup>	38.59	9,338
Gypsum	kg y <sup>-1</sup>	NA	NA
Peat Moss	kg y <sup>-1</sup>	25.11	1,792(1412)
Irrigation Water (Farm Land)	kg y <sup>-1</sup>	6.00 <sup>a</sup>	95(35.9)
Irrigation Water (Nursery)	kg y <sup>-1</sup>	6.00 <sup>a</sup>	33(2.72)
Seeds	kg y <sup>-1</sup>	30	40.0(31.5)
Total C derived from photosynthesis	kg y <sup>-1</sup>	NA	3,944(3107)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	35.2	2,817(2220)
Amananth	kg y <sup>-1</sup>	33.5	32(37)
Amananth, Red	kg y <sup>-1</sup>	34.9	13(19)
Choy Sum	kg y <sup>-1</sup>	32.6	1,773(1781)
Lettuce	kg y <sup>-1</sup>	35.9	6(16)
Nai Bai	kg y <sup>-1</sup>	34.1	17(22)
New Zealand Spinach	kg y <sup>-1</sup>	38.7	649(167)
Sweet Potato Leaf	kg y <sup>-1</sup>	39.7	179(113)
Water Spinach	kg y <sup>-1</sup>	36.2	655(314)
Xiao Pak Choy	kg y <sup>-1</sup>	31.5	113(152)
C Monoxide	g min <sup>-1</sup>	NA	0.0084(0.0069)
C Dioxide	g min <sup>-1</sup>	NA	4.82(6.68)
Plastic Waste (Farm land)	kg y <sup>-1</sup>	1.24	49.2
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	166.2(38.4)

<sup>a</sup> C concentration in mg per litre (mg L<sup>-1</sup>)

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. C concentration of runoff is 14.43 mg L<sup>-1</sup> and leaching is 17.14 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket)

NA: Not available

The photosynthetic C input in CF1 contributed 3,803 kg C ha<sup>-1</sup> y<sup>-1</sup> which accounted for 33% of total C input (Figure 4.6). The photosynthetic C was also one of the main C contributors in OF1 and OF2 that resulted in 1,193  $\pm$  506 and 3,944  $\pm$  3,107 kg C ha<sup>-1</sup> y<sup>-1</sup> input.

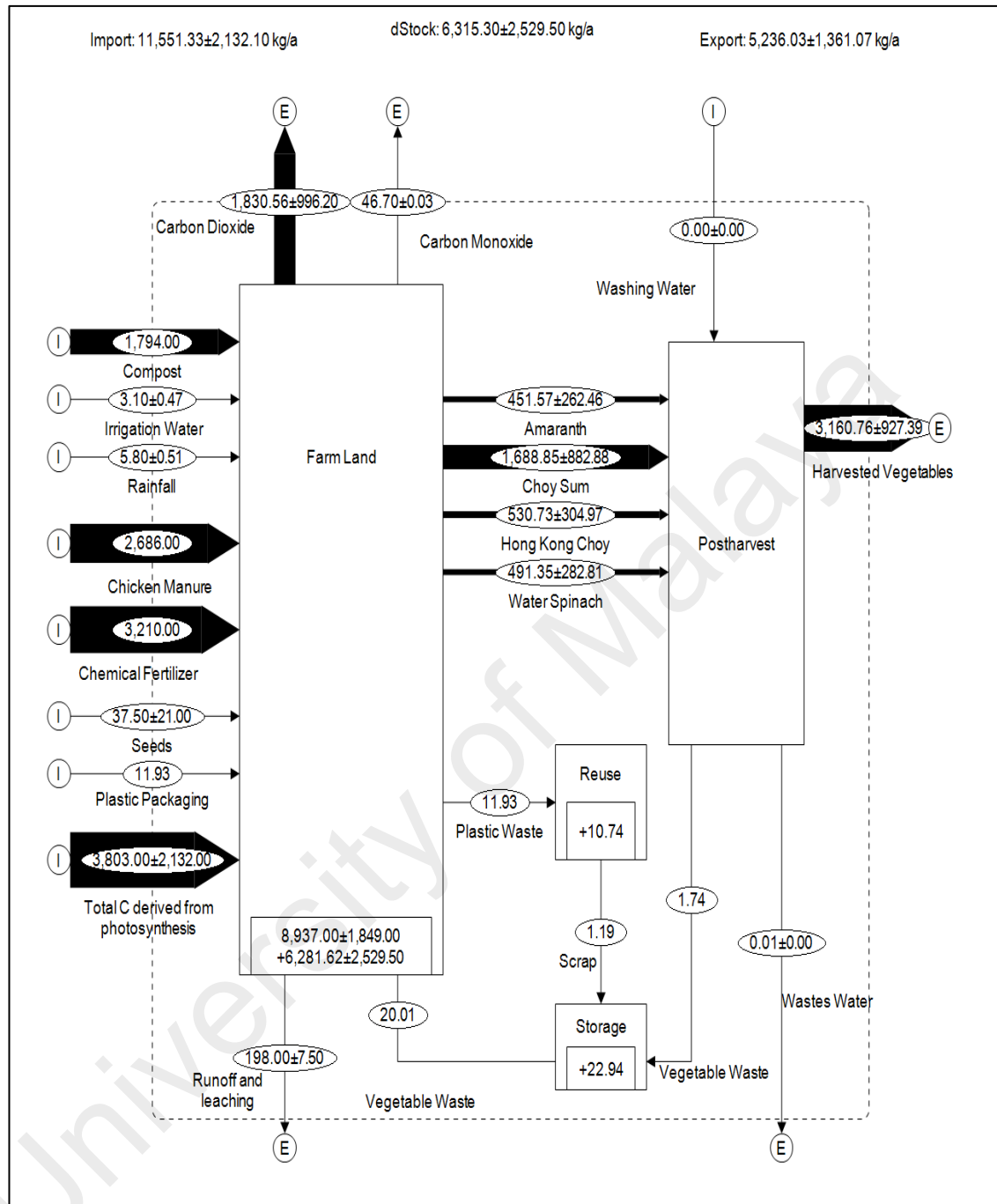


Figure 4.6: C flow in conventional farm CF1 ( $\text{kg C ha}^{-1} \text{y}^{-1}$ )

However, about  $358 \pm 40 \text{ kg C ha}^{-1} \text{y}^{-1}$  of total photosynthetic C was recorded for CF2 which is much lower as compared to other farms in this study (Figure 4.7). The low annual crop yield in CF2 leads to less amount of photosynthetic C. Study of vegetable farms in Afghanistan showed for two years the total photosynthetic C input was 9,321

$\pm 1,503 \text{ kg C ha}^{-1}$ , thus on average about  $4,000 \text{ kg C ha}^{-1}$  enters the farm systems yearly (Safi *et al.*, 2011b). The photosynthetic C of CF1 and OF2 showed comparable results to Safi *et al.* (2011b), but CF2 and OF2 showed lower photosynthetic C input. Total C derived from photosynthesis is defined as the C in root exudates, death roots and a certain fraction of leaf litter that were left on the field (Scotti *et al.*, 2015a). Thus, it is estimated based on crop yield as it is correlated with crop residues amount. This also means the differences of the photosynthetic C volume varied according to crop yield of each farm and were influenced by farm management, soil properties, nutrient management, climate, crop types and etc.

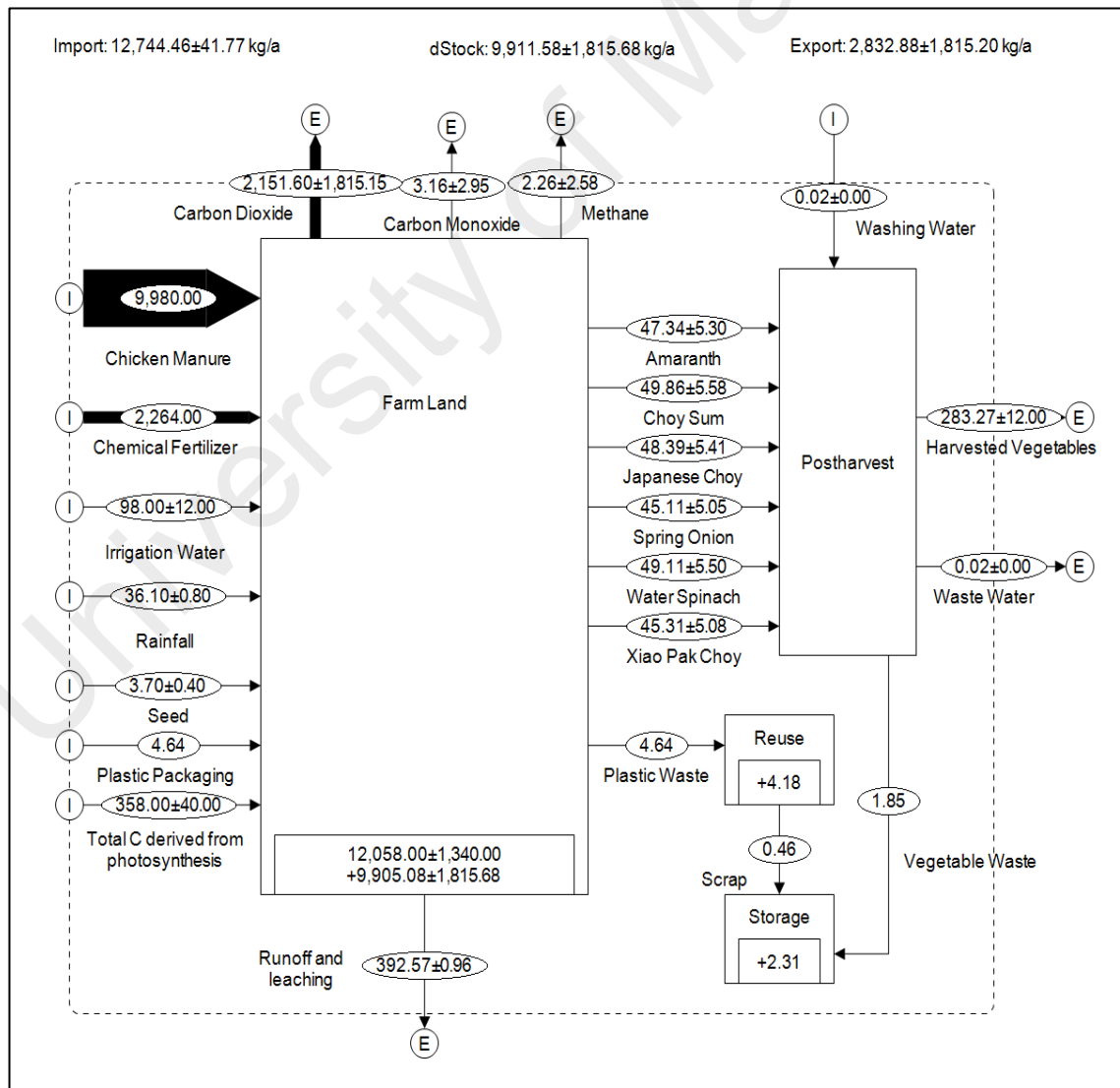


Figure 4.7: C flow in conventional farm CF2 ( $\text{kg C ha}^{-1} \text{ y}^{-1}$ )

Certification requirements defined allowable nutrient sources used in organic farming systems (Abbott & Manning, 2015). Hence, organic matter such as compost is the primary nutrient source especially in organic systems. Similarly, compost has become the major C contributor in organic systems. Results showed compost application accounts for 93% (18,680 kg C ha<sup>-1</sup> y<sup>-1</sup>) and 71% (11,871 kg C ha<sup>-1</sup> y<sup>-1</sup>) of the total C input in OF1 and OF2 (Figures 4.8 and 4.9).

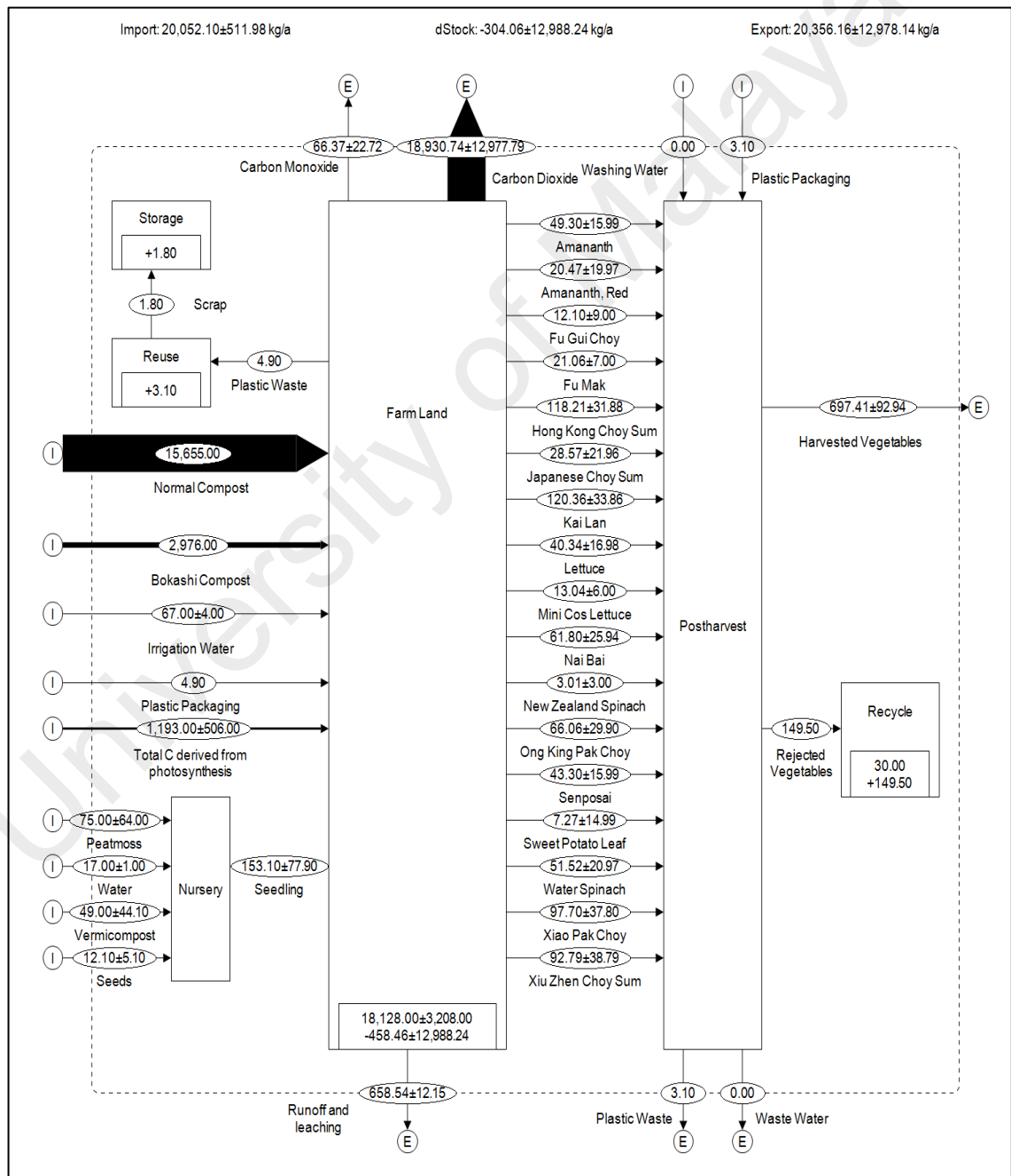


Figure 4.8: C flow in organic farm OF1 (kg C ha<sup>-1</sup> y<sup>-1</sup>)

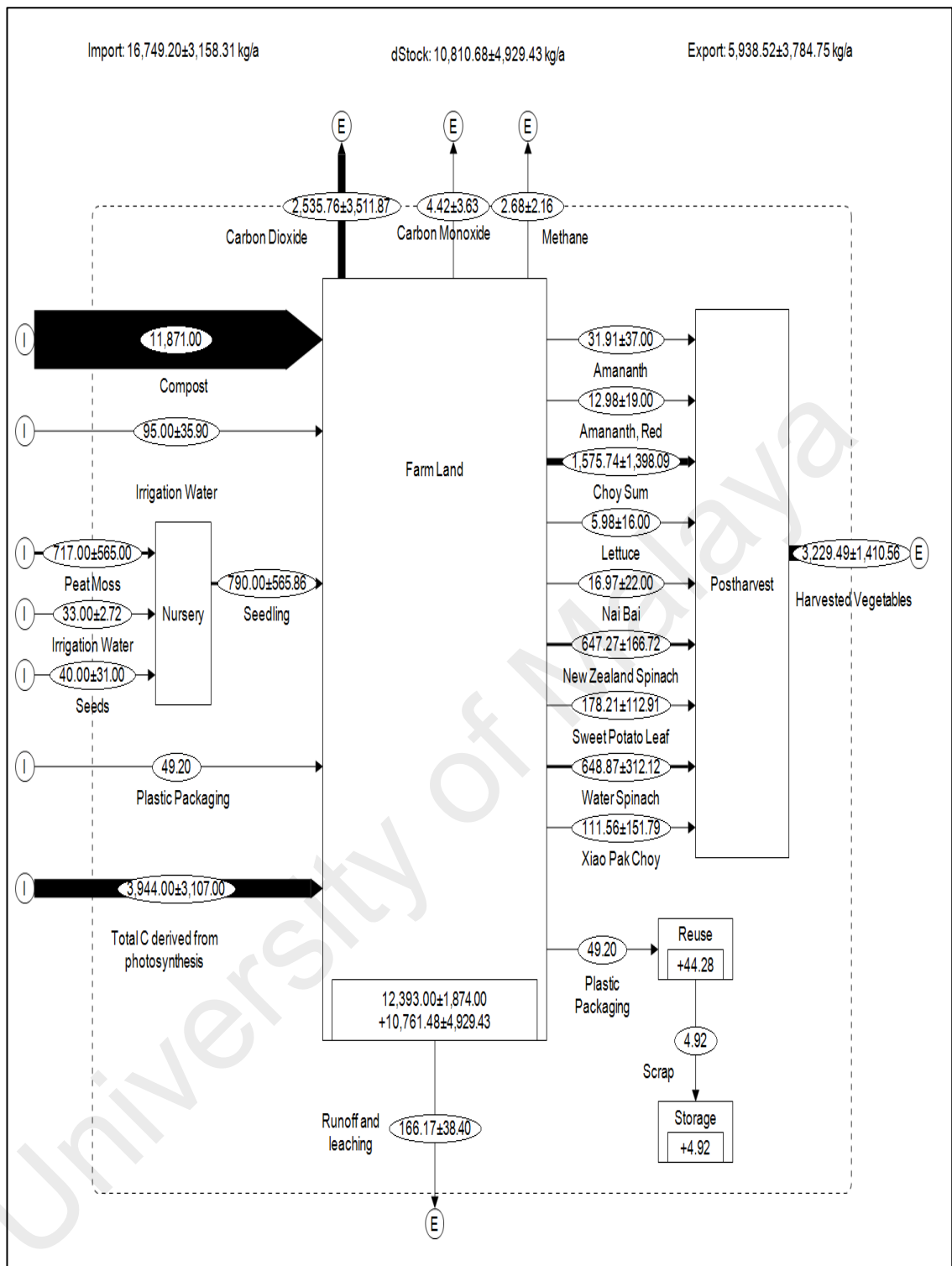


Figure 4.9: C flow in organic farm OF2 ( $\text{kg C ha}^{-1} \text{y}^{-1}$ )

Other than that, compost application also resulted  $1,794 \text{ kg C ha}^{-1} \text{y}^{-1}$  input in CF1 which is around 16% of the total C. A study using isotope of C ( $^{13}\text{C}$ ) to trace the soil C flow further supports the majority of increase in soil C is derived from the compost and

biomass input (Jaiarree *et al.*, 2014). The C content of the compost used by the farms in this study varies due to different composting sources and materials. The C content in compost used in CF1 and OF2 were 36.10% and 38.59% which were slightly higher than Ksheem *et al.* which was 28.52% (Ksheem *et al.*, 2015). Three types of compost were used in OF1 and were generally contained lower C content as compared to CF1 and OF2. Normal compost, Bokashi compost and vermicompost were recorded with total C concentrations of 11.79, 22.42 and 12.03%, respectively. A number of authors have reported the differences in C concentrations of composts were due to variations in compost material, maturity, quality, and composting period (C content decreases as compost progress along time) (Barje *et al.*, 2013; García *et al.*, 1993; Goyal *et al.*, 2005; Zmora-Nahum *et al.*, 2005).

The use of chicken manure is very common in conventional farm systems and it is one of the major organic matter input (Liao *et al.*, 2015). Chicken manure was one of the main C input in conventional systems that accounts for 23 % (2,666 kg C ha<sup>-1</sup> y<sup>-1</sup>) and 78 % (9,980 kg C ha<sup>-1</sup> y<sup>-1</sup>) of C input in CF1 and CF2. The result is supported by Jia *et al.* (2012) who indicated that about 23-73% of C entered into agroecosystems primarily through manure application. Study in China showed that the manure applied in greenhouse vegetable cultivation soil contributed to 5,800 kg C ha<sup>-1</sup> y<sup>-1</sup> which is higher than the results of CF1 and lower than CF2 (Wang *et al.*, 2011). The manure applications rate was lower in CF1 as compared to CF2 and this might be due to the compost inputs in CF1 has supplemented the required amount of nutrients. The chicken manure used in CF1 and CF2 contained C concentrations of 20.44% and 24.93% which were similar with the results reported by Jia *et al.* (2012) that chicken manure contained 21.1% of C. In addition, sampling of aged manures in a study also showed total C ranged from 26.8 to 29.2% (Hartz *et al.*, 2000).

Peat moss usage was only observed at organic systems in nursery as potting material for seeds growth before transplanting to “Farmland”. The peat moss input accounts for 0.4% ( $75 \pm 64 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) and 4.2% ( $717 \pm 565 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) of total C input in OF1 and OF2. Chemical fertilizer often assumed to only provide nutrient to crop growth. However, the C analysis reveals the inert material in filler of fertilizer contained about 38.5% of C (Tables 4.6 and 4.7). Thus, large quantity of chemical fertilizer application especially under intensive farming will result in a considerable amount of C input into farm systems. The chemical fertilizer applications in CF1 and CF2 have resulted with 3,210 and 2,264  $\text{kg C ha}^{-1} \text{ y}^{-1}$  mass input which contributed 28% and 18% of total C, respectively. However, the C input of mineral fertilizers was considered insignificant in study by Wu et al. (2015) as it only accounted for about 2% of the total C inputs. The differences in results might be due to the type of chemical fertilizer used in the farm. The fertilizer used in Wu et al. (2015) is urea which is a type of straight fertilizer (0.2% C concentrations). On the other hand, the fertilizer in this study was compound fertilizer (38.5% C concentrations). This is supported by Otero et al. (2005) who highlighted that straight fertilizer contained only one primary nutrient while compound fertilizer usually contained N, P, K and filler. The function of filler in fertilizer was to provide bulk, prevent caking, and provide essential nutrient and it is often made of ground limestone, phosphogypsum, or other inert material depends on manufacturer (UN, 1998). Most of the research on C flow analysis, C balance or budget in farm systems do not include the C input of chemical fertilizer and peat moss due to the general assumption of low C contribution of farm C flux (Goenster *et al.*, 2014; Jia *et al.*, 2012a; Liao *et al.*, 2015; Wang *et al.*, 2011).

Irrigation water, rainfall, seeds and plastic packaging have contributed around 1% or less of C input in all the farms in this study. Water input via irrigation and rainfall



contained low concentrations of C, therefore resulted in small amount of C input even though the volume of water flow was high. The C input in CF1, CF2, OF1, and OF2 through irrigation was  $3.1 \pm 0.47$ ,  $98 \pm 12$ ,  $84 \pm 5$ ,  $128 \pm 38.62$  kg C ha<sup>-1</sup> y<sup>-1</sup> with C concentrations of 0.579, 9.91, 3.47, and 6.0 mgL<sup>-1</sup> (Figure 4.10), respectively. A study of low input and high input vegetable home gardens indicated low C flux of 1 and 29 kg C ha<sup>-1</sup> y<sup>-1</sup> by irrigation (Goenster *et al.*, 2014). The much lower irrigation volume of 1,304 m<sup>3</sup> ha<sup>-1</sup> y<sup>-1</sup> (20 mgL<sup>-1</sup> C concentrations) in the study by Goenster *et al.* (2014) as compared to the irrigation rate in this study (5,000-20,000 m<sup>3</sup> ha<sup>-1</sup> y<sup>-1</sup>) explained the variation of the results. Some studies showed higher C contribution from irrigation that is about 10% of total C input ( $235.5 \pm 105.9$  kg C ha<sup>-1</sup> for two years  $\approx 117.8$  kg C ha<sup>-1</sup> y<sup>-1</sup>) (Safi, 2011; Safi *et al.*, 2011a). Different results reported might be due to the variation of irrigation sources (Abdalla *et al.*, 2012). The irrigations used in Safi *et al.* (2011a) were from the mixed of well, river and sewage sludge which contained high C concentrations while in this study the farm's main water sources was from catchment pond.

Direct rainfall input was only observed in conventional systems of CF1 and CF2 which accounts for  $5.80 \pm 0.51$  and  $36.1 \pm 0.80$  kg C ha<sup>-1</sup> y<sup>-1</sup>. The results were lower as compared to Goenster *et al.* (2014) who reported 143 kg C ha<sup>-1</sup> y<sup>-1</sup> flux in farms located at tropical region. The average rainfall volume in CF1 and CF2 range from 1,382 to 1,782 mm and the C content in rainfall samples were 0.222 and 1.59 mgL<sup>-1</sup> (Figure 4.11). Even though, the rainfall volume in farms in this study was higher than Goenster *et al.* (2014) but the higher C concentrations (17.5 mg L<sup>-1</sup>) resulted higher rainfall C flow. On the other hand, the rainfall C input in CF2 was similar with the results of Safi (2011) that recorded 36.95 kg C ha<sup>-1</sup> y<sup>-1</sup> ( $73.9 \pm 9.1$  for two years). The rainfall C input of Safi (2011) was similar to CF2 however the rainfall volume was much lower (176-346 mm)

as it located at arid to semiarid climate region. This suggest higher C concentration in rainfall samples of Safi (2011), The results highlighted the variation of rainfall volume and C content were the main factors influencing the C influx through rainfall.

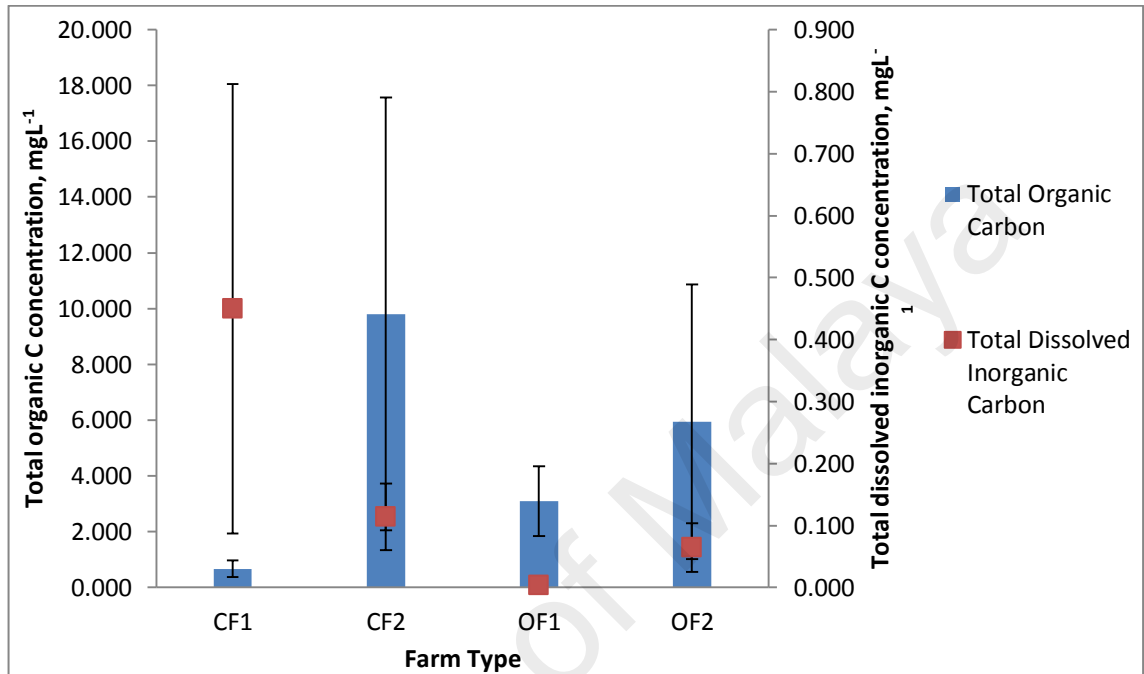


Figure 4.10: TOC and inorganic C concentrations in irrigation

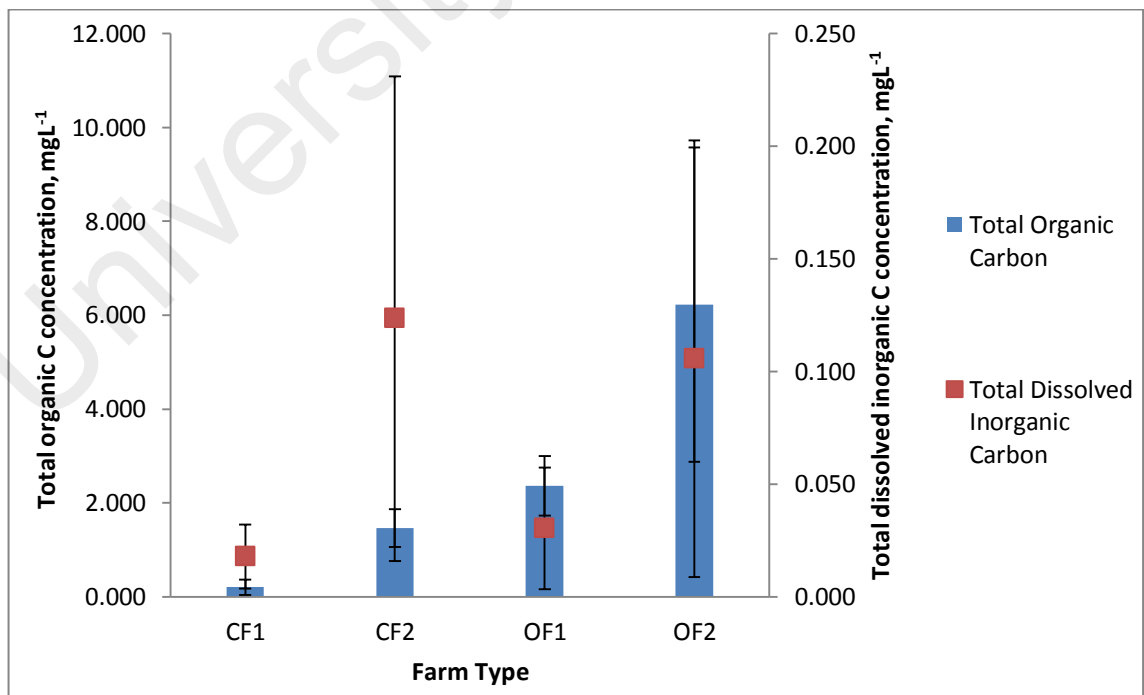


Figure 4.11: TOC and inorganic C concentrations in rainfall

There were three primary C output routes which included harvested vegetables, gaseous emissions, and runoff and leaching from the farm systems. Vegetable harvesting was the main C output from CF1 ( $3,161 \pm 927 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) and OF2 ( $3,229 \pm 1,410 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ). The C output of vegetable harvesting in of CF1 and OF2 were similar to results of a two year study on four different cropping systems whereby the average harvested vegetable C output were 3096, 4194, 4019, and 2768 kg C ha<sup>-1</sup> y<sup>-1</sup> (Jia *et al.*, 2012a). A study by Wang *et al.* (2011) has recorded higher C output of vegetable harvesting ranging between 5,430 and 8,850 kg C ha<sup>-1</sup> y<sup>-1</sup> (Clark *et al.*, 1998; Wang *et al.*, 2011). The C analysis demonstrates low variation of C content (range from 34.14% to 36.2%) in each type of leafy vegetables cultivating in the farms in this study (Appendix A). This suggests the main factor for differences in harvested vegetable C output was farm yield instead of C content in vegetables. Thus, the low harvested vegetable C output in CF2 ( $283 \pm 12 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) and OF1 ( $697 \pm 93 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) were mainly due to low vegetable yields.

The C flow indicated gaseous emissions was one of the primary C mass output and CO<sub>2</sub>, CO, and CH<sub>4</sub> were the three major gaseous (Refer to Section 4.82, 4.83 and 4.84 for details). Among the three measured gaseous, CO<sub>2</sub> emissions recorded the highest in all the farms in this study. The CO<sub>2</sub> emissions in CF1, CF2, OF1 and OF2 resulted C mass output of  $1,831 \pm 996$ ,  $2,152 \pm 1,815$ ,  $18,931 \pm 12,978$ , and  $2,536 \pm 3,512 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . This was supported by Siegfried *et al.* (2011) who reported >98% of the C gaseous losses were from CO<sub>2</sub>. The C mass loss of CO emissions only accounts for 0.4% ( $47 \pm 0.03 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ), 0.02% ( $3.16 \pm 2.95 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ), 0.33% ( $66.37 \pm 22.72 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ), and 0.02% ( $4.42 \pm 3.63 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) of total C output in CF1, CF2, OF1 and OF2, respectively.

Methane emissions were only observed in CF2 and OF2 with  $2.26 \pm 2.58$  and  $2.68 \pm 2.16$  kg C ha<sup>-1</sup> y<sup>-1</sup> of C mass flux. Overall, the total C output via gaseous emissions from CF1, CF2, OF1 and OF2 were 1,878, 2,157, 18,997, and 2,543 kg C ha<sup>-1</sup> y<sup>-1</sup>, respectively. A study of intensive vegetable farming in Southern China indicated the C emissions range from 3,507 to 10,757 kg C ha<sup>-1</sup> y<sup>-1</sup> (Mu *et al.*, 2013). Similarly, an earlier study reported high C gaseous losses were from organic systems which emit 6,200, 9,700 and 10,600 kg C ha<sup>-1</sup> y<sup>-1</sup> (Siegfried *et al.*, 2011). The average gaseous C emission in OF1 was 7-10 times higher as compared to other farm systems in this study. However, a much higher result was reported by Predotova *et al.* (2010) where about 26,000 kg C ha<sup>-1</sup> y<sup>-1</sup> of C flux from urban and peri-urban intensive vegetable farms were observed. The author further conclude the high CO<sub>2</sub> fluxes from the farms were due to high intensity vegetable cropping, high organic matter input, short cultivation periods, fast turnover under the warm and moist conditions microclimate (Predotova *et al.*, 2010). The degradation of organic inputs in the soil surface emits CO<sub>2</sub> under aerobic condition while CH<sub>4</sub> is produced especially under anoxic condition (Scotti *et al.*, 2015a). In addition, tillage regime in farms may increase soil gaseous exchange between different soils layers and promotes the mineralization of organic C. Therefore, the variations of the C emissions might be due to various factors such as soil types (composition, temperature, respiration and moisture), organic matters input, soil microbial dynamic and tillage practices (Akiyama *et al.*, 2014; Elder & Lal, 2008; Sainju *et al.*, 2012; Schlesinger & Andrews, 2000; Tiwari *et al.*, 2015).

The material flows indicated that water was the major input-output in the farm systems and C analysis of runoff and leachate shows the water discharged did transport the C away from the farm system. Based on water flow model by Bengtsson *et al.* (2003), the potential water that may leave the farm system as runoff and leaching can be estimated.

With the assumption that the water output was 50% via runoff and 50% via leaching the C mass output were estimated. The potential C mass loss through runoff and leaching in CF1, CF2, OF1 and OF2 was about  $198 \pm 7.5$ ,  $393 \pm 0.96$ ,  $659 \pm 12$  and  $166 \pm 38$  kg C  $\text{ha}^{-1} \text{y}^{-1}$ , respectively. Basically the C mass lost through runoff and leaching were low and contributed to 1.7%, 3%, 3.2% and 1% of C lost in CF1, CF2, OF1 and OF2, respectively. The C loss via water route was insignificant as compared to the C output of vegetable harvesting and gaseous emissions. Both CF2 and OF1 has higher amount of manure and compost inputs as compared to CF1 and OF2 which might be the reason for high C lost in runoff and leaching. This was supported by Evanylo et al. (2008) who demonstrated that the C content in runoff of soil amended with compost and manure were higher than soil with and without mineral fertilizer application. Since the C lost through runoff and leaching was based on the assumption of 50% lost through runoff and 50% lost through leaching thus were likely to be overestimated or underestimated. Even though in the material flow indicated high volume of water flow within the farm systems, the C lost was generally low due to average low C concentrations. It is observed that the C lost through runoff and leaching in conventional farm systems CF1 and CF2 were sensitive to the amount and timing of rainfall; hence the results only reflect the condition during the study period (Chou *et al.*, 2008). Despite the fact that C was lost from the farm systems, the C that leaches away may enters the water table or surface water which indirectly contributed to C sequestration (Nordt *et al.*, 2000). However, there is a possibility that excessive C input to aquatic system may lead to detrimental effect on the ecosystem (Tuvendal & Elmqvist, 2011).

Overall, the average C input in CF1, CF2, OF1 and OF2 was  $11,551 \pm 2,132$ ,  $12,744 \pm 42$ ,  $20,052 \pm 512$  and  $16,749 \pm 3,158$  kg C  $\text{ha}^{-1} \text{y}^{-1}$ , respectively. The results observed were comparable to urban and peri-urban vegetable farms in Niamey, Niger which the

total C input in high input farms averaged at around 30,000 kg C ha<sup>-1</sup> y<sup>-1</sup> whereas in low input farms it was 7,000 kg C ha<sup>-1</sup> y<sup>-1</sup> (Diogo *et al.*, 2010). However, a study using RothC model showed the estimated total C input of major crop types ranged from 910 kg C ha<sup>-1</sup> y<sup>-1</sup> to 6,780 kg C ha<sup>-1</sup> y<sup>-1</sup> which is lower than this study (Meersmans *et al.*, 2013). The variation from the observed results mainly due to different methods used in C input quantification. The total C exports from the farm systems was recorded as 5,236 ± 1,361, 2,833 ± 1,815, 20,356 ± 12,978 and 5,938 ± 3,785 kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2, OF1 and OF2, respectively. The C output of CF1, CF2, and OF2 was lower than a study on vegetable home gardens which the average C output of 9,634 and 8,473 kg C ha<sup>-1</sup> y<sup>-1</sup> for high input and low input gardens (Goenster *et al.*, 2014).

The STAN software performed data reconciliation and validation on the mean value and standard deviation to eliminate data contradictions with the assumption that the data were normally distributed (Cencic & Rechberger, 2008). Accordingly, the data presented in C balance generated by STAN were adjusted in order to resolve the contradiction. Based on the STAN model, the annual C balances was around 6,315 ± 2,529, 9,912 ± 1,816 and 10,802 ± 4,929 kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2 and OF2, respectively. The C balance of CF2 and OF2 were comparable to Diogo *et al.* (2010) who recorded 9,936 kg C ha<sup>-1</sup> y<sup>-1</sup> of C balance in urban and peri-urban vegetables farms. In addition, the result of CF1 was similar to the finding of Abdalla *et al.* (2012) who recorded 6,412 kg C ha<sup>-1</sup> y<sup>-1</sup> of C balance in vegetables home gardens. The positive stock change of the C flow models in CF1, CF2 and OF2 suggested the farm systems have the potential to be a C sink (Kellner *et al.*, 2011). Large amount of organic matters input such as compost and C derived from photosynthesis (crop residues) have contributed significant amount of C input into the farm systems.

The organic system OF1 is the only farm in this study that showed a potential source of C with C balance of  $-304 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . The values was lower as compared to the research done by Vleeshouwers and Verhagen (2002) in which  $-840 \text{ kg C ha}^{-1} \text{ y}^{-1}$  of C exited conventional managed arable land but the result was higher than Goenster et al. (2014) who recorded a C balance of  $-21 \text{ kg C ha}^{-1} \text{ y}^{-1}$  in home gardens at Nuba Mountains, Sudan. Study by Abbott & Manning (2015) has justified that the organic system if under a particular environments may lead to 'mining' or reduction of existing nutrient resources. The C flow indicated that the C deficit in OF1 mainly due to high gaseous emissions of  $\text{CO}_2$ . The high variation of  $\text{CO}_2$  emission causes higher standard deviation of C balance of  $12,988 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . Gaseous emissions exceeding 100% variability were often observed in field due to complex link of factors that influenced flux rate (Bashir *et al.*, 2013; Hénault *et al.*, 2012; Sainju *et al.*, 2012). The C gaseous emissions in OF1 recorded highest among the farms in this study with total lost around  $19,000 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . The high C gaseous emissions in OF1 might be due to the high C input of  $20,052 \pm 512 \text{ kg C ha}^{-1} \text{ y}^{-1}$  which is  $4000\text{-}8000 \text{ kg C ha}^{-1} \text{ y}^{-1}$  higher than other farm systems. This was supported by Janzen (2006) who suggested that the high C input increased soil C and encouraged soil microbial decomposition activity that leads to increased C gaseous emissions (Janzen, 2006). In addition, studies highlighted that the soil C accumulations and sequestration relied on factors that limit the decomposers/microbial (eg. temperature) instead of large organic matter input (Jarecki & Lal, 2003; Lal, 2004b; Schlesinger & Andrews, 2000). This suggests the soil condition in OF1 might favour the microbial decomposition activity and increases C gaseous emissions which lead to C deficit. Crop yield and gaseous emissions were the main C outflows in the farm systems while the major C contributor in the farm systems was organic matter inputs such as manure, compost and photosynthetic C (crop residues). The preferable option to increase farm C stocks without compromising the

farm production was to increase organic matters input or minimize C gaseous emissions. However, there were reports questioning the benefits of increased C input for C sequestration because increased of soil C may induce C gaseous emissions (Janzen, 2006). In this study, the total C input of 20,000 kg C ha<sup>-1</sup> y<sup>-1</sup> in OF1 has resulted in high C gaseous emissions that leads to C deficit in the farm system. In contrast the total C input ranged between 11,000-17,000 kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2 and OF2 which contributed to C stock in the systems. Therefore, there is a possibility of C input threshold that may lead to high C emissions once it is exceeded. However, further study is required to determine the correlation between soil C input, microbial activity and C gaseous emissions.

The C flow model implies that both organic and conventional systems do have the potential to be C sink even when C emissions were taken into account in the C flow analysis. The C flux model shows that there is still need for improvements in OF1 through proper nutrient management. Study suggested that organic matters applications improved soil C pool more significantly than inorganic fertilizer (Gregorich *et al.*, 2001). However, constant monitoring is required to ensure C input rate is able to counterbalance the C gaseous emissions loss. The results also highlighted the C sequestration contribution from manure and crop residues in conventional farms. Thus, intensive agriculture with high yield might contribute to considerable amount of C sequestration in agriculture soil.

#### **4.4 N flow analysis (STAN model)**

N in soil can easily transform from one form to another form and high deposition of N in soil often leads to losses that impact the environment (Galloway *et al.*, 2008; Vitousek *et al.*, 1997). The main objective of N flow analysis is to assess potential N



surplus in the farm system. A positive nutrient balance indicates a potential loss of nutrients to the environment or nutrient accumulation in soil, whereas a negative balance signifies soil nutrient depletion (Bouwman *et al.*, 2013). The N flow analysis is a practical and low cost method that identifies and estimate potential impact or unsustainable use of resources (Parris, 1998). All the N flux relevant processes and materials were monitored and quantified during the study period and then modeled with STAN 2.5 software. Based on the site survey and STAN model, the N flux metabolism models of each farm system were generated and expressed as shown in Eq. 4.9-4.12.

$$NS_{CF1} = (C_{nm} + CM_{nm} + IW_{nm} + R_{nm} + CF_{nm} + S_{nm} + BNF + PP_{nm} + W_{nm}) - (HV_{nm} + G_{nm} + WW_{nm}) \quad (\text{Eq. 4.9})$$

$$NS_{CF2} = (CM_{nm} + IW_{nm} + R_{nm} + CF_{nm} + S_{nm} + BNF + PP_{nm} + W_{nm}) - (HV_{nm} + G_{nm} + WW_{nm}) \quad (\text{Eq. 4.10})$$

$$NS_{OF1} = (C_{nm} + BC_{nm} + VC_{nm} + IW_{nm} + S_{nm} + BNF + PM_{nm} + PP_{nm} + W_{nm}) - (HV_{nm} + G_{nm} + WW_{nm} + PW_{nm}) \quad (\text{Eq. 4.11})$$

$$NS_{OF2} = (C_{nm} + S_{nm} + PM_{nm} + IW_{nm} + BNF + PP_{nm}) - (HV_{nm} + G_{nm}) \quad (\text{Eq. 4.12})$$

Where, NS is N surplus of CF1, CF2, OF1 and OF2 (N surplus that may lost via volatilization, denitrification, leaching, runoff or stored in soil);  $C_{nm}$ , is N mass of compost;  $BC_{nm}$ , is N mass of Bokashi compost;  $VC_{nm}$ , is N mass of vermicompost;  $CM_{nm}$ , is N mass of chicken manure;  $PM_{nm}$ , is N mass of peat moss;  $IW_{nm}$ , is N mass of irrigation water;  $R_{nm}$ , is N mass of rainfall;  $S_{nm}$ , is N mass of seed; BNF, biological N fixation;  $PP_{nm}$ , is N mass of plastic packaging;  $W_{nm}$ , is N mass of

washing water; HV, nm, is N mass of harvested vegetable; G, nm, is N mass of gaseous emissions (ammonia and nitrous oxide); WW, nm, is N mass of waste water; PW, nm, is N mass of plastic packaging wastes. Table 4.10 show the input-output tabulation of N mass and the N concentrations for the various commodities in CF1. The key N input in CF1 was from chemical fertilizer, however, compost and chicken manure also contribute considerable amount of N. The main N outputs in CF1 were runoff, leaching and vegetable production.

Table 4.10: N input-output in CF1

<b>Input</b>	<b>Unit</b>	<b>Average N Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) N per Hectare<sup>c</sup></b>
Compost	kg y <sup>-1</sup>	4.199	209
Chicken Manure	kg y <sup>-1</sup>	2.610	343
Chemical Fertilizer	kg y <sup>-1</sup>	16	1,334
Irrigation Water	kg y <sup>-1</sup>	9.62 <sup>a</sup>	50.9(9.2)
Rain	kg y <sup>-1</sup>	2.24 <sup>a</sup>	58.4(3.5)
Washing Water	kg y <sup>-1</sup>	9.62 <sup>a</sup>	0.0486(0.0029)
Seeds	kg y <sup>-1</sup>	6	7.5(4.2)
Estimated N fixation	kg y <sup>-1</sup>	NA	6.11(2.70)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	4.735	355(199)
Amaranth	kg y <sup>-1</sup>	3.946	53(30)
Choy Sum	kg y <sup>-1</sup>	5.037	254(126)
Hong Kong Choy Sum	kg y <sup>-1</sup>	5.526	85(43)
Water Spinach	kg y <sup>-1</sup>	4.429	61(35)
Ammonia	g min <sup>-1</sup>	NA	0.215(0.107)
Nitrous Oxide	kg y <sup>-1</sup>	NA	26.11(1.246)
Plastic Waste (Farm land)	kg y <sup>-1</sup>	0.10	0.962
Vegetable Wastes (Farm Land)	kg y <sup>-1</sup>	4.735	2.62
Vegetable Wastes (Postharvest)	kg y <sup>-1</sup>	4.735	0.227
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	716(0.94)
Waste Water (Postharvest)	kg y <sup>-1</sup>	9.62 <sup>a</sup>	0.0486(0.0029)

<sup>a</sup> N concentration in mg per litre (mg L<sup>-1</sup>)

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. N concentration of runoff is 45.1 mg L<sup>-1</sup> and leaching is 7.3 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket)

NA: Not available

Table 4.11 listed all the main N input and output in CF2. Similar to CF1, chemical fertilizer and chicken manure were the major N contributor. Runoff and leaching was the main N output route in CF2.

Table 4.11: N input-output in CF2

<b>Input</b>	<b>Unit</b>	<b>Average N Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) N per Hectare <sup>c</sup></b>
Chicken Manure	kg y <sup>-1</sup>	1.43	572
Chemical Fertilizer	kg y <sup>-1</sup>	16	941
Irrigation Water	kg y <sup>-1</sup>	10.35 <sup>a</sup>	102.3(7.2)
Rain	kg y <sup>-1</sup>	1.13 <sup>a</sup>	25.7(1.1)
Washing Water (Postharvest)	kg y <sup>-1</sup>	10.35 <sup>a</sup>	0.0190(0.0009)
Seeds	kg y <sup>-1</sup>	6	0.7(0.1)
Estimated N fixation	kg y <sup>-1</sup>	NA	6.22(2.45)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	5.024	38(4)
Amaranth	kg y <sup>-1</sup>	5.132	7.32(0.81)
Choy Sum	kg y <sup>-1</sup>	6.080	9.75(0.97)
Japanese Choy Sum	kg y <sup>-1</sup>	4.869	7.81(0.77)
Spring Onion	kg y <sup>-1</sup>	3.996	4.99(0.63)
Water Spinach	kg y <sup>-1</sup>	4.180	5.96(0.66)
Xiao Pak Choy	kg y <sup>-1</sup>	5.888	8.39(0.93)
Ammonia	g min <sup>-1</sup>	NA	0.0058(0.0056)
Nitrous Oxide	kg y <sup>-1</sup>	NA	21.6(1.14)
Plastic Waste (Farm Land)	kg y <sup>-1</sup>	0.10	0.3743
Vegetable Wastes ( Postharvest )	kg y <sup>-1</sup>	5.024	0.2723
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	1,474(11.5)
Waste Water ( Postharvest )	kg y <sup>-1</sup>	10.35 <sup>a</sup>	0.0190(0.0009)

<sup>a</sup> N concentration in mg per litre (mg L<sup>-1</sup>)

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. N concentration of runoff is 51.5 mg L<sup>-1</sup> and leaching is 82.9 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket)

NA: Not available

The main N input and output in OF1 were listed in Table 4.12. Compost and irrigation were the main N input in OF1. Similar to other farms, the vegetable production in OF1 contributed considerable amount of N output. Runoff and leaching was one of the main N output in OF1. The N concentration of runoff was 86.5 mgL<sup>-1</sup> and leaching was 249.7 mgL<sup>-1</sup> in OF1. This was higher than other farms in this study.

Table 4.12: N input-output in OF1

Input	Unit	Average N Concentration, %	Total Average ( $\pm$ SD) N per Hectare <sup>c</sup>
Normal Compost	kg y <sup>-1</sup>	1.456	1933
Bokashi Compost	kg y <sup>-1</sup>	1.987	264
Vermicompost	kg y <sup>-1</sup>	1.092	4.64
Peat Moss	kg y <sup>-1</sup>	0.275	5.94(2.5)
Irrigation Water (Farm Land)	kg y <sup>-1</sup>	21.15 <sup>a</sup>	406.2(5.5)
Water (Nursery)	kg y <sup>-1</sup>	21.15 <sup>a</sup>	102.4(0.8)
Washing Water (Postharvest)	kg y <sup>-1</sup>	21.15 <sup>a</sup>	0.0024(0.0)
Seeds	kg y <sup>-1</sup>	6	2.4(1.0)
Estimated N fixation	kg y <sup>-1</sup>	NA	4.89(2.27)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	5.15	114(48)
Amananth	kg y <sup>-1</sup>	5.376	7.78(2.59)
Amananth, Red	kg y <sup>-1</sup>	3.993	2.308(2.59)
Fu Gui Choy	kg y <sup>-1</sup>	5.523	2.13(2.58)
Fu Mak	kg y <sup>-1</sup>	2.700	1.403(0.48)
Hong Kong Choy Sum	kg y <sup>-1</sup>	4.810	18.15(4.39)
Japanese Choy Sum	kg y <sup>-1</sup>	5.779	4.51(3.63)
Kai Lan	kg y <sup>-1</sup>	5.720	20.18(4.64)
Lettuce	kg y <sup>-1</sup>	4.654	5.14(4.36)
Mini Cos Lettuce	kg y <sup>-1</sup>	4.915	1.80(1.79)
Nai Bai	kg y <sup>-1</sup>	6.275	11.17(7.77)
New Zealand Spinach	kg y <sup>-1</sup>	4.450	0.368(0.31)
Ong King Pak Choy	kg y <sup>-1</sup>	6.206	11.70(0.31)
Senposai	kg y <sup>-1</sup>	4.689	5.80(4.19)
Sweet Potato Leaf	kg y <sup>-1</sup>	6.082	1.032(2.03)
Water Spinach	kg y <sup>-1</sup>	4.943	7.18(2.91)
Xiao Pak Choy	kg y <sup>-1</sup>	6.332	16.76(6.64)
Xiu Zhen Choy Sum	kg y <sup>-1</sup>	5.133	14.03(5.38)
Ammonia	g min <sup>-1</sup>	NA	0.2027(0.0312)
Nitrous Oxide	kg y <sup>-1</sup>	NA	35.04(1.495)
Rejected Vegetables	kg y <sup>-1</sup>	5.15	21.9
Plastic Wastes (Postharvest)	kg y <sup>-1</sup>	0.1	0.246
Plastic Wastes (Farm Land)	kg y <sup>-1</sup>	0.1	0.39
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	2,257(36.4)
Waste Water (Postharvest)	kg y <sup>-1</sup>	21.15 <sup>a</sup>	0.0024(0.00)

<sup>a</sup> N concentration in mg per litre (mg L<sup>-1</sup>)

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. N concentration of runoff is 86.5 mg L<sup>-1</sup> and leaching is 249.7 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket)

NA: Not available

Table 4.12 listed all the main N input and output in OF2. Similar to OF1, compost was the main N input in OF2. In addition, irrigation water also contributed considerable amount of N. Vegetable production, runoff and leaching were the main N output in OF2.

Table 4.13: N input-output in OF2

<b>Input</b>	<b>Unit</b>	<b>Average N Concentration, %</b>	<b>Total Average (<math>\pm</math> SD) N per Hectare <sup>c</sup></b>
Compost (Midori 333)	kg y <sup>-1</sup>	3.368	815
Gypsum	kg y <sup>-1</sup>	NA	NA
Peat Moss	kg y <sup>-1</sup>	0.392	28(22)
Irrigation Water (Farm Land)	kg y <sup>-1</sup>	24.06	379(111)
Water (Nursery)	kg y <sup>-1</sup>	24.06	132(8)
Seeds	kg y <sup>-1</sup>	6	8.0(6.3)
Estimated N fixation	kg y <sup>-1</sup>	NA	4.48(3.19)
<b>Output</b>			
Total Vegetable Production	kg y <sup>-1</sup>	4.846	372(285)
Amananth	kg y <sup>-1</sup>	5.376	5.198(5.855)
Amananth, Red	kg y <sup>-1</sup>	3.993	1.445(2.155)
Choy Sum	kg y <sup>-1</sup>	5.104	277.2(247.6)
Lettuce	kg y <sup>-1</sup>	4.654	0.804(4.232)
Nai Bai	kg y <sup>-1</sup>	6.275	3.165(6.588)
New Zealand Spinach	kg y <sup>-1</sup>	4.450	74.63(17.10)
Sweet Potato Leaf	kg y <sup>-1</sup>	3.063	13.85(7.782)
Water Spinach	kg y <sup>-1</sup>	5.200	93.95(45.10)
Xiao Pak Choy	kg y <sup>-1</sup>	5.495	19.80(26.55)
Ammonia	g min <sup>-1</sup>	NA	0.0060(0.0056)
Nitrous Oxide	kg y <sup>-1</sup>	NA	18.1(2.8)
Plastic Waste (Farm Land)	kg y <sup>-1</sup>	0.1000	3.97
Runoff and leaching	kg y <sup>-1</sup>	<sup>b</sup>	1,332(533)

<sup>a</sup> N concentration in mg per litre (mg L<sup>-1</sup>)

<sup>b</sup> Total water output is assumed to be 50% via runoff and 50% via leaching. N concentration of runoff is 77.4 mg L<sup>-1</sup> and leaching is 214.0 mg L<sup>-1</sup>.

<sup>c</sup> SD: standard deviation (value in bracket)

NA: Not available

Chemical fertilizer and chicken manure were the two main N inputs in conventional farm systems of this study (Figure 4.12 and 4.13). Chemical fertilizer used in CF1 and CF2 accounts for 1,334 and 941 kg N ha<sup>-1</sup> y<sup>-1</sup> respectively, while chicken manure contributed 343 and 572 kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively. Similar N input was reported in a

survey of 56 greenhouse vegetable farms in China which demonstrated the average total N inputs of chemical fertilizer and manure were 1,358 and 1,881 kg N ha<sup>-1</sup> y<sup>-1</sup> (Ju *et al.*, 2006).

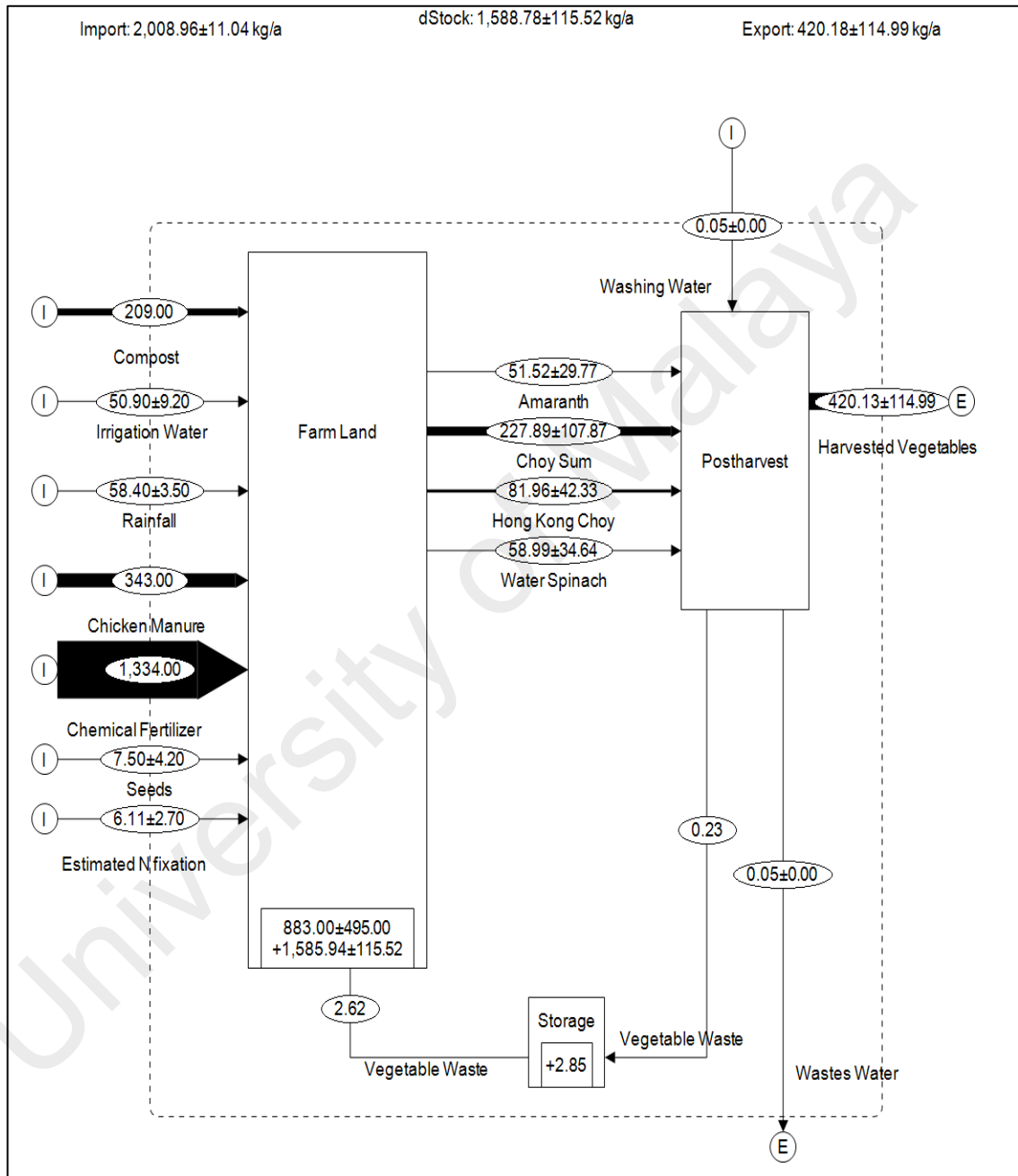


Figure 4.12: N flow in conventional farm CF1 (kg N ha<sup>-1</sup> y<sup>-1</sup>)

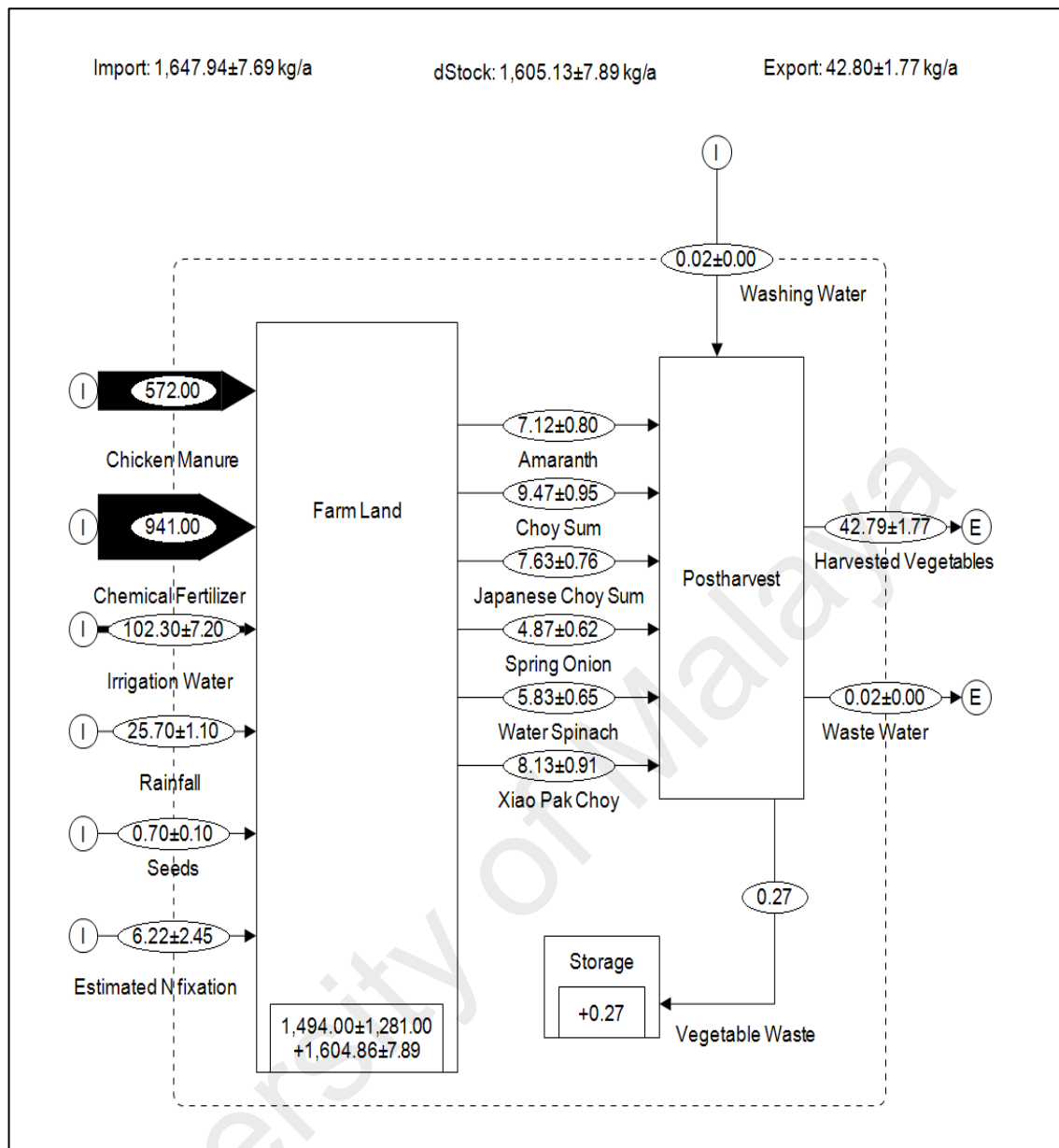


Figure 4.13: N flow in conventional farm CF2 ( $\text{kg N ha}^{-1} \text{y}^{-1}$ )

A study also showed similar results in which 78% of the surveyed farms applied N (organic manure and chemical fertilizers) at a rate range from 300 to 900  $\text{kg N ha}^{-1} \text{y}^{-1}$  and more than 35% of the surveyed farms received total N greater than 1000  $\text{kg N ha}^{-1} \text{y}^{-1}$  (Chen *et al.*, 2004). The manure application rate in this study was low as compared to Huang *et al.* (2006) which recorded 510-3,600  $\text{kg N ha}^{-1} \text{y}^{-1}$  of N input via manure (Huang *et al.*, 2006a). However, the results were similar to Abdulkadir *et al.* (2013) who reported that the manure input resulted in 650-856  $\text{kg N ha}^{-1} \text{y}^{-1}$  input. The manure

applications rates vary among farm systems and the N content in the manure is also one of the variation factors. The N content in chicken manure of this study was similar with poultry manure used in Lim and Vimala (2012) which was around 1.80%. However, the N content of manure reported by Chen et al. (2004) was much higher which range from 38.9-58.7%.

The primary N input in OF1 and OF2 was compost, which accounts for 81% (2,201 kg N ha<sup>-1</sup> year<sup>-1</sup>) and 60% (815 kg N ha<sup>-1</sup> year<sup>-1</sup>) of the total N input (Figure 4.14 and 4.15), respectively. However, in CF1 the compost was also applied as soil amendments which contributed around 209 kg N ha<sup>-1</sup> y<sup>-1</sup> input which accounts for 10% of total N. The compost N content reported by Raviv et al. (2004) was between 2.39% and 2.84% which were lower than the compost used in CF1 (4.2%) and OF2 (3.37%) but higher than OF1 (1.09-1.99%).

The N content of compost differs among the farms in this study might be due to different compost quality which were affected by sources, composition and batches. The composts used in the farms in this study generally have C:N ratio of 11:1. The lower C:N ratio signified higher tendency of N mineralization to available N for crop use. However, the mineralization rate was highly dependable on the soil microbial activities (Gaskell & Smith, 2007). According to OF1 grower, high amount of compost was applied in order to achieve desirable yield because it is the sole nutrient source for crop growth. This was supported by Wong et al. (1999) and Lim & Vimala (2012) which reported organic nutrient application rate may require 10 to 30 folds of inorganic fertilizer amount in order to achieve good yield (Lim & Vimala, 2012; Wong *et al.*, 1999).



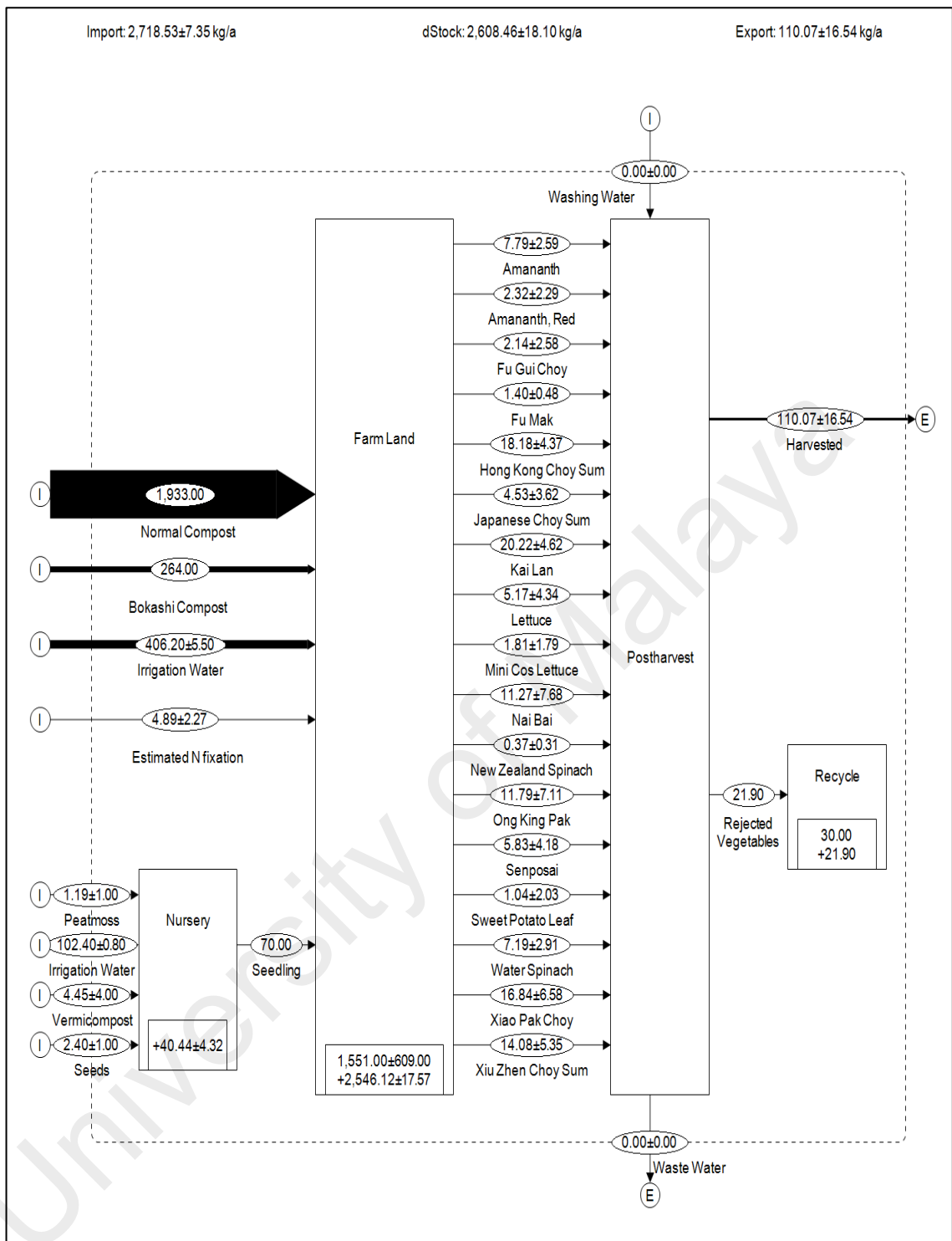


Figure 4.14: N flow in organic farm OF1 ( $\text{kg N ha}^{-1} \text{y}^{-1}$ )

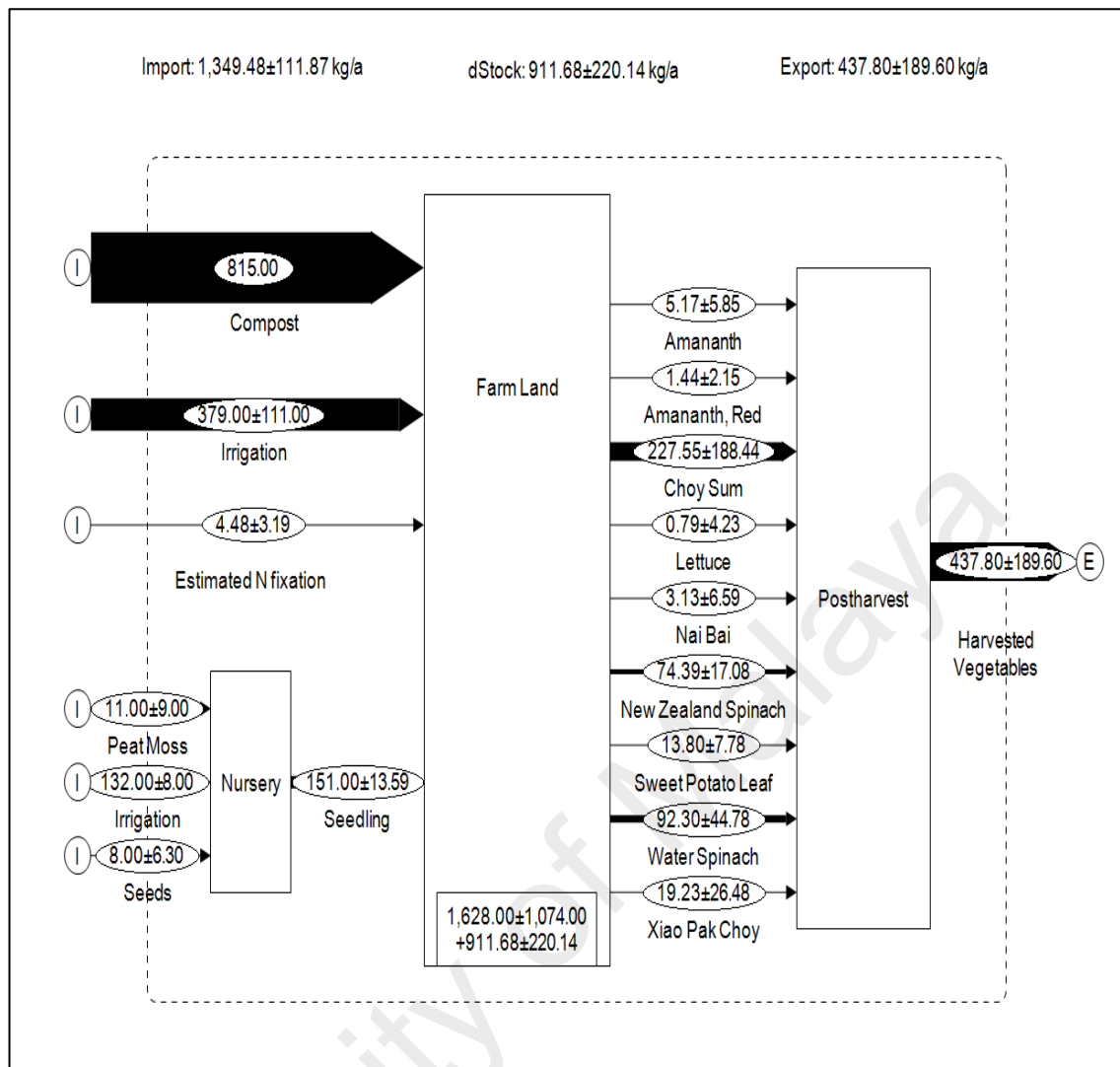


Figure 4.15: N flow in organic farm OF2 ( $\text{kg N ha}^{-1} \text{y}^{-1}$ )

The intense irrigation activities in CF1, CF2, OF1, and OF2 resulted in significant N inputs of  $51 \pm 9.2$ ,  $102 \pm 7.2$ ,  $508 \pm 6.3$ , and  $511 \pm 119 \text{ kg N ha}^{-1} \text{y}^{-1}$ , respectively. The N input via irrigation has contributed to about 19% and 37% of total N input in OF1 and OF2. Comparable results were reported in Guimera (1998) where the irrigation input carried 15% to 50% of the total N applied into the horticulture and flower crop systems. Similarly, in Song et al. (2009) and Ju et al. (2006) the irrigation water in greenhouse vegetables farms contributed N input of 59-603 and 402  $\text{kg N ha}^{-1} \text{y}^{-1}$ , respectively. Several authors showed different results of irrigation N input which range at 1-32  $\text{kg N ha}^{-1} \text{y}^{-1}$  (Goenster *et al.*, 2014; Zhang *et al.*, 2015). Different results observed might due

to the variations in irrigation systems, irrigation rates and N concentrations in the irrigation water. The average N concentrations of irrigation water samples from CF1, CF2, OF1 and OF2 was about 9.62, 10.35, 21.15, and 24.06 mg L<sup>-1</sup>, respectively. Similar results were observed in Goenster et al. (2014) with N concentrations 27 mg L<sup>-1</sup> in irrigation water (Figure 4.16). The results were aligned with Song et al. (2009), in which the total N concentrations range at 5-82 mg L<sup>-1</sup>. The N content in irrigation was highly dependent on the water source (eg. catchment pond, underground water, well, river, and wastewater). Studies conducted by Ju et al. (2006), Khai et al. (2007), Diogo et al. (2010), and Min et al. (2012) highlighted the high degrees of influences of irrigation sources on N fluxes.

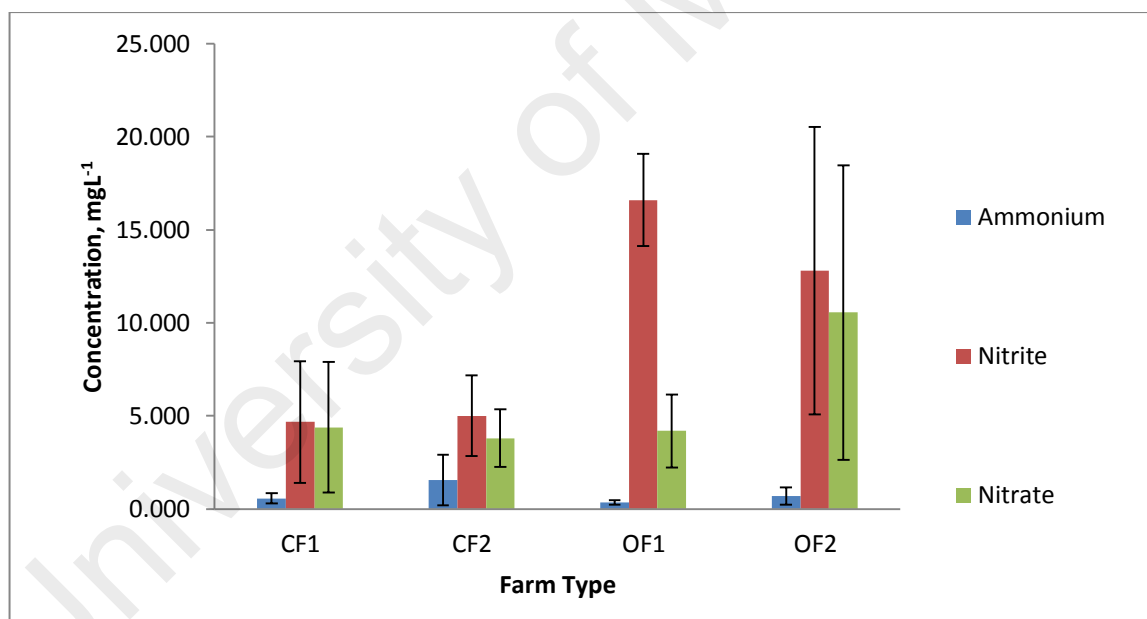


Figure 4.16: Ammonium, nitrite, and nitrate concentrations in irrigation water

Conventional farms have a lower irrigation volume than organic farm systems because the former employed open field cultivation. In this case, rainfall serves as the main water source in the farms. In present works, the rainfall in CF1 and CF2 resulted in N inputs of  $58 \pm 3.5$  and  $26 \pm 1.1$  kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively. The results were higher than Ju et al. (2006) where the rainfall resulted in N input of 14.2 and 18.9 kg N ha<sup>-1</sup> y<sup>-1</sup> but

similar to Zhang et al. (2015) that recorded rainfall N input of 15 and 30 kg N ha<sup>-1</sup> y<sup>-1</sup>. The amount of N input via rainfall highly depends on climate, thus explains the variation of results in different studies. This study was conducted in tropical country-Malaysia which has an average rainfall volume of 2000 mm every year. The high volume of rainfall into farms in this study might result in higher N flux through rain-fed as compared to studies that were located at temperate regions. Example in country such as Norway, the low average rainfall of 600 mm y<sup>-1</sup> only contributes 6.20 ± 0.40 kg N ha<sup>-1</sup> y<sup>-1</sup> flux in the farms (Korsaeth, 2008). The rainfall N concentrations in CF1 (2.24 mgL<sup>-1</sup>) and CF2 (1.13 mgL<sup>-1</sup>) were slightly higher as compared to Goenster et al. (2014) who recorded N concentrations of 0.7 mgL<sup>-1</sup> which resulted in N input of 6 kg N ha<sup>-1</sup> y<sup>-1</sup> (818mm rainfall).

The analysis of rainfall in all farms in this study showed variations of N concentrations (Figure 4.17). This signified that the condition of each farm is different even though they were located in the same climate region. Research showed the main N source of rainfall was from volatilization of ammonia from farmland (Wang *et al.*, 2004). Thus, higher N content in rainfall in OF1 and OF2 indicated higher ammonia volatilization in the farm areas. Generally, variation in rainfall N concentrations and rainfall volumes were the two factors affecting total rainfall N flux.

Biological N fixation (BNF) estimated for CF1, CF2, OF1 and OF2 was 6.11 ± 2.70, 6.22 ± 2.45, 4.89 ± 2.27 and 4.48 ± 3.19 kg N ha<sup>-1</sup> y<sup>-1</sup> respectively, which generally contributed to less than one percent of total N input. Similar estimation done by Abdulkadir et al. (2013) reported an average BNF of 3-7 kg N ha<sup>-1</sup> y<sup>-1</sup> (Abdulkadir *et al.*, 2013). However, a study showed a wider range of N input 0-55.2 kg N ha<sup>-1</sup> y<sup>-1</sup> from BNF (Korsaeth, 2008). The estimated BNF in conventional systems (CF1 and CF2)

were slightly higher than organic systems (OF1 and OF2). This was supported by a comparative study of farms in China, Brazil and Egypt which showed that BNF in conventional systems was 1.2 times higher than organic systems (Oelofse *et al.*, 2010). The differences were mainly due to variations in microbial activities, efficiencies of N use, soil conditions (e.g high C:N ratio limits BNF), soil temperature (e.g., *Azospirillum* species thrive in more in tropical environments), and crop types (host to rhizosphere or photosynthesis bacteria) (de Bruijn, 2015; Wagner, 2011).

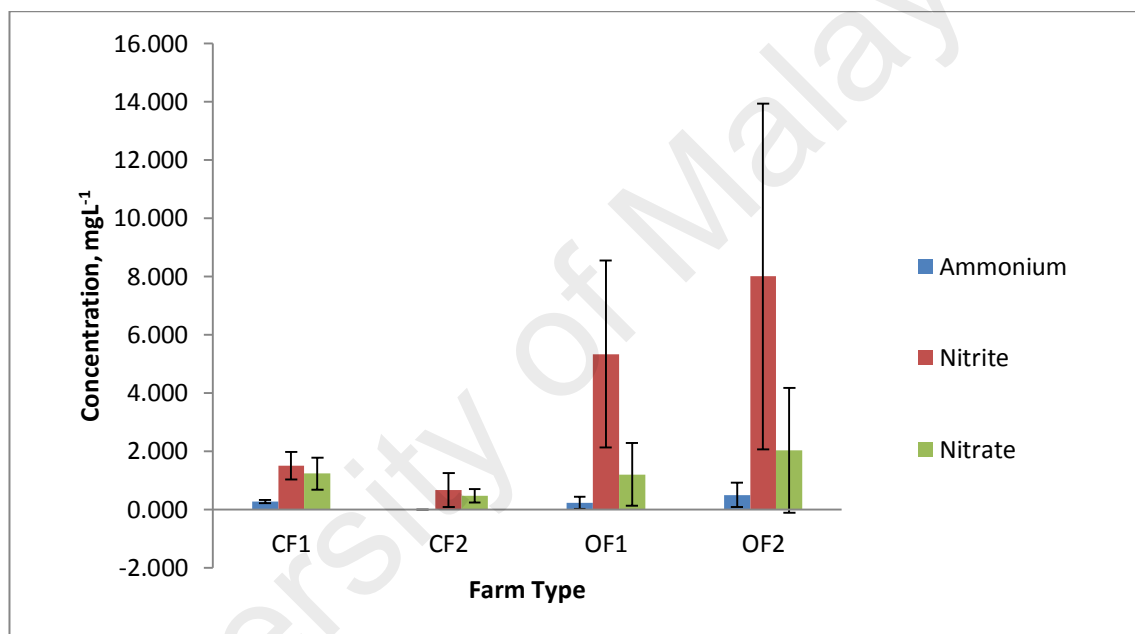


Figure 4.17: Ammonium, nitrite, and nitrate concentrations in rainfall

Other inputs such as peat moss and seeds account for the rest of the N input (< 1% of total N input). The peat moss usage was observed only under “Nursery” process in organic farm systems which resulted in  $1.19 \pm 1.0$  and  $11.0 \pm 9.0$  kg N ha<sup>-1</sup> y<sup>-1</sup> of input in OF1 and OF2. The total N contribution of seeds was  $7.5 \pm 4.2$ ,  $0.7 \pm 0.1$ ,  $2.4 \pm 1.0$  and  $8.0 \pm 6.3$  kg N ha<sup>-1</sup> y<sup>-1</sup> respectively. Comparable results were reported by Korsaeath (2008) that the seed N input range from 2.5 to 10.2 kg N ha<sup>-1</sup> y<sup>-1</sup>. The N contribution of peat moss and seeds were insignificant as compared to the N input from fertilizer and

compost. Several authors had excluded the N input from seeds and peatmoss due to the low quantities (Abdulkadir *et al.*, 2013; Goenster *et al.*, 2014; Ju *et al.*, 2006).

Vegetable harvesting was the main N output from the farms in this study which resulted  $420 \pm 115$ ,  $43 \pm 2$ ,  $110 \pm 17$  and  $438 \pm 190$  kg N ha<sup>-1</sup> y<sup>-1</sup> output in CF1, CF2, OF1 and OF2, respectively. The results of CF1 and OF2 were in line with Ju *et al.* (2006), Ju *et al.* (2007) and Abdulkadir *et al.* (2013) whereby the N removal of plant harvesting contributed N output range from 90 to 500 kg N ha<sup>-1</sup> y<sup>-1</sup>. The N analysis of each variety of vegetables in each farm showed little variation in vegetables N content (4.74-5.15%) (Appendix B). Therefore, the main factor for variation in vegetable N output was crop yield.

The N inputs in organic farms were often perceived to be low because they depend on organic inputs, biological fixation, crop rotation, nutrient cycling of crop residues, and nutrient retention via green manure to meet nutrient demand for plant growth (ADAS *et al.*, 2006; Lynch *et al.*, 2012). However, the results in this study showed otherwise. No significant differences ( $P > 0.05$ ) were observed among the annual N inputs between organic and conventional systems and similar results were also reported by Korsath (2008). The amount of N input was highest in OF1 followed by CF1, CF2, and OF2. The high N input at OF1 was mainly due to the intensive usage of compost which was about 206 t ha<sup>-1</sup> y<sup>-1</sup>. This value was equated to a total N flux of 2,201 kg N ha<sup>-1</sup> y<sup>-1</sup> into the farm. The volume of compost used in OF1 was six times higher than the volume used in OF2. However, study suggested that the total N input via organic matters were most likely immobilized in the soil environment as the soil available N for crop growth relied on microbial mineralization which was influenced by soil temperature and moisture (Gaskell & Smith, 2007).

The STAN software generated N balance model by calculation and reconciliation of the N input-output data. According to the model, all the farms could be categorized as N “sink” given the N balances of  $1,589 \pm 116$ ,  $1,605 \pm 8$ ,  $2,608 \pm 18$ , and  $912 \pm 220$  kg N  $\text{ha}^{-1} \text{y}^{-1}$  in CF1, CF2, OF1, and OF2, respectively. Several authors reported lower N balances that were less than  $1,000$  kg N  $\text{ha}^{-1} \text{y}^{-1}$  (Huang *et al.*, 2006a; Khai *et al.*, 2007; Korsath, 2008; Salo & Turtola, 2006; Wang *et al.*, 2008). However, other farms reported N balances up to 3,327 and 4,328 kg N  $\text{ha}^{-1} \text{y}^{-1}$  (Ju *et al.*, 2006; Ju *et al.*, 2007). The variations in the N balances of farm systems were most likely caused by irregular fertilizer applications (Wang *et al.*, 2008). In addition, the N surplus might be due to the variations in N availability of manure and compost applied in this study. This was supported by Gaskell & Smith (2007) which highlighted the high variability of available N release from organic materials and the study also indicated N recovery from organic materials for subsequent cropping was low (4-15%). The quantity and timing of soil N availability also influenced the N balance especially in vegetable production system due to the relatively short growing cycle (Rosen & Allan, 2007). Therefore, the uncertainty of N availability in soil and unsynchronized timing of N availability and crop growth might be the reason for N accumulation and over fertilization in the farms in this study. The N surplus and over fertilized soil can affect crop growth and cause nutrient loss that impact the environment.

Farmers' awareness on the importance of nutrient management and plant nutrient were crucial for sustainable development in agriculture sector. However, the interview from this study suggested that the farm owners of CF2 and OF1 lack of such knowledge. Farmer from CF2 revealed that he was not aware of the significance N inputs from manure and irrigation application. On the other hand, the grower from OF1 assumed

that crop yield can be boosted by increasing compost application. Unfortunately, an increase in N application does not necessarily improve crop yields (Ju *et al.*, 2006). Crops require a certain amount of nutrient for optimal growth. Thus, excess nutrients applied are exposed to runoff and leaching (DEFRA, 2015b).

The positive stock change in this study indicated N surpluses which potentially accumulate in various soil fractions or become lost to the environment via denitrification, runoff, or leaching (Bouwman *et al.*, 2013; Ju *et al.*, 2006). However, the nutrients leaving agriculture lands do not always enter water bodies; they may be retained by soil particles along the transport route (Bashir *et al.*, 2013; Reidsma *et al.*, 2012). Therefore, transport factors such as soil texture, soil permeability, cation exchange capacity, saturated hydraulic conductivity, slope of cultivation area, distance of agriculture land to water body, riparian buffer zone, irrigation erosion, rainfall, surface runoff, leaching, soil erosion, drainage, and biological activity will determine the extent of nutrients entering water bodies (Nie *et al.*, 2012; Pärn *et al.*, 2012; Venohr *et al.*, 2011).

#### **4.5 Nutrient budget of C and N**

The C and N flow in previous section was modelled with STAN software which takes into consideration of the uncertainty value (standard deviation) by performing data validation and reconciliation. In this section, the C and N balances were tabulated with nutrient budgeting (also known as nutrient accounting) which was often used in nutrient balance study (Abdulkadir *et al.*, 2013; Bengtsson *et al.*, 2003; Brouwer, 1998; DEFRA, 2014b; Goenster *et al.*, 2014; Ju *et al.*, 2011; Ju *et al.*, 2006; Lazzerini *et al.*, 2014; Roy *et al.*, 2003; Salo & Turtola, 2006; Scoones & Toulmin, 1998; Shober *et al.*, 2011;



Wortmann & Kaizzi, 1998). The nutrient budget employs simple accounting of input-output calculation of data.

The C and N budgets in organic and conventional farms are presented in Figure 4.18. The average annual C in conventional farm systems CF1 and CF2 were +6,943 and +6,943 kg C ha<sup>-1</sup> y<sup>-1</sup> while in OF1 and OF2 were +530 and +9,940 kg C ha<sup>-1</sup> y<sup>-1</sup>. All the farms in this study have positive C balances which were aligned with the C flows generated by STAN software (Section 4.3), except for OF1. Interestingly, the STAN generated C flow model for OF1 indicated deficit of C stock in the farm which were contradicting with the results of the nutrient budgets.

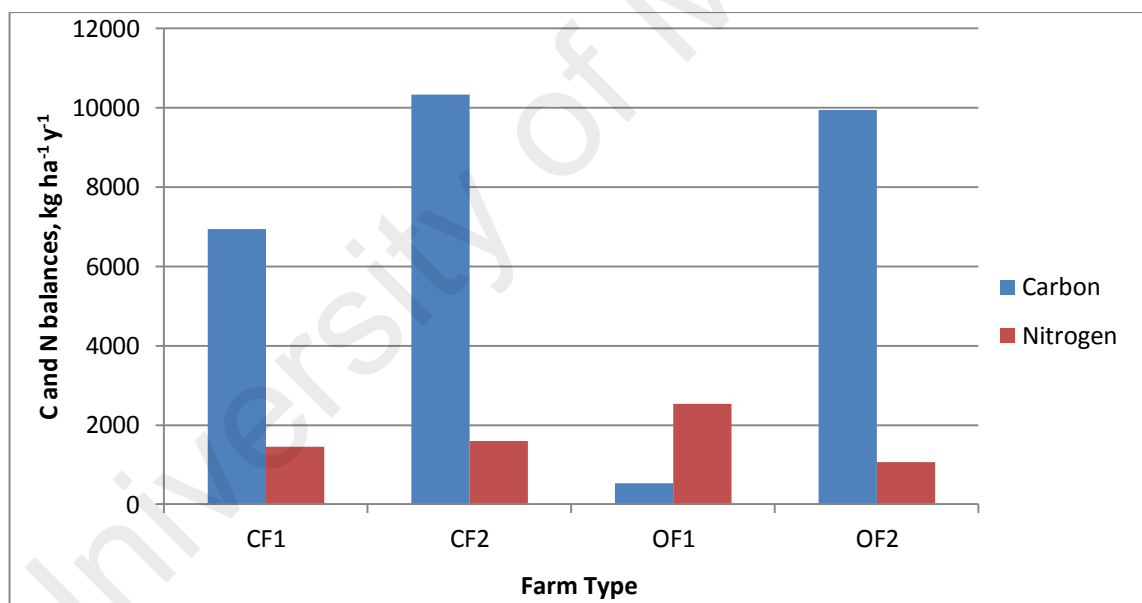


Figure 4.18: Annual balance of C and N, kg ha<sup>-1</sup> year<sup>-1</sup>

The average annual N balances in CF1, CF2, OF1 and OF2 tabulated with nutrient budgeting were +1,517 and +1,589, +2,547 and +1,059 kg N ha<sup>-1</sup> y<sup>-1</sup>. The results were aligned with the N flow generated by STAN software (Section 4.4), whereby all the farms in this study have N surplus. The results indicated no significant differences in N surplus between organic and conventional farm systems. Similar results were observed

in case studies in China, Brazil and Egypt which certified organic and non-organic farms have similar nutrient surplus magnitude (Oelofse *et al.*, 2010). Surplus nutrient budget indicated both the study organic and conventional farms were not depleting soil reserves, however, large surplus in long term may lead to nutrient loss that impact the environment.

The annual total C inputs tabulated with nutrient budgeting were about 11,540, 12,740, 20,382 and 15,297 kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2, OF1 and OF2 (Figure 4.19), respectively. On the other hand, the annual total N inputs tabulated were 2,009, 1,648 2,805 and 1,449 kg N ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2, OF1 and OF2 (Figure 4.20), respectively. Compost input was the major sources of C and N in organic systems which accounts for 61-92% and 56-79% of total C and N inputs. In contrast, the major C input in conventional system was chicken manure which contributed 23% and 79% of total C in CF1 and CF2, respectively.

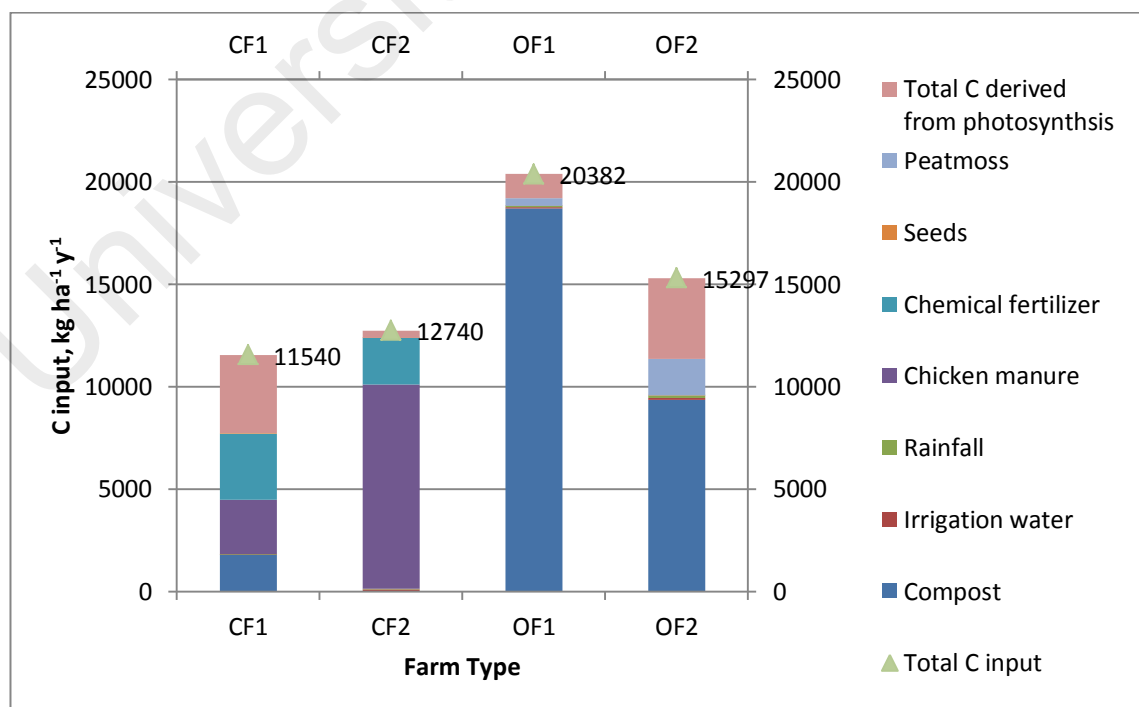


Figure 4.19: Annual cumulative input fluxes of C, kg ha<sup>-1</sup> year<sup>-1</sup>

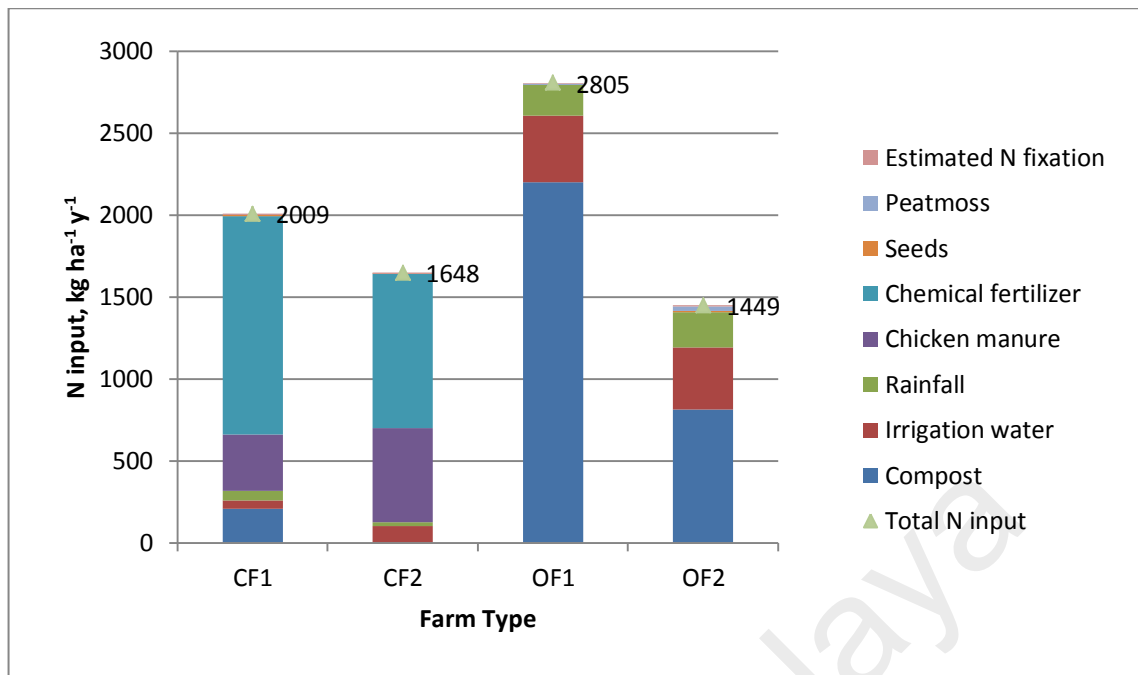


Figure 4.20: Annual cumulative input fluxes of N,  $\text{kg ha}^{-1} \text{ year}^{-1}$

Chemical fertilizer application was the major contributor of N input in CF1 and CF2 which accounts for 66% and 57% of total N. Photosynthetic C contributed considerable amount of C input in all study farm. Nutrient budgeting estimated the total C outputs in CF1, CF2, OF1 and OF2 was about 4,597, 2,406, 19,852 and 5,357  $\text{kg C ha}^{-1} \text{ y}^{-1}$ , respectively (Figure 4.21) while total N outputs were around 491, 59, 259 and 390  $\text{kg N ha}^{-1} \text{ y}^{-1}$ , respectively (Figure 4.22). There is a large export of C from both farm systems via crop harvesting and  $\text{CO}_2$  emissions. High  $\text{CO}_2$  emissions were observed at OF1 and CF2 which accounts for 95% and 80% of total C output, respectively. In contrast, crop harvesting was high in CF1 and OF2 which accounts for 59% and 52% of total C output, respectively. Harvested vegetable was the main N output in all the farms in this study which accounts for 44-95% of total N output. The variation of results between STAN flow model and nutrient budgeting might be due to the different methodology approaches, data reconciliation and standard deviation. Studies also indicated different nutrient budgeting methods resulted in different nutrient balance results (Oenema *et al.*, 2003; Watson & Atkinson, 1999).

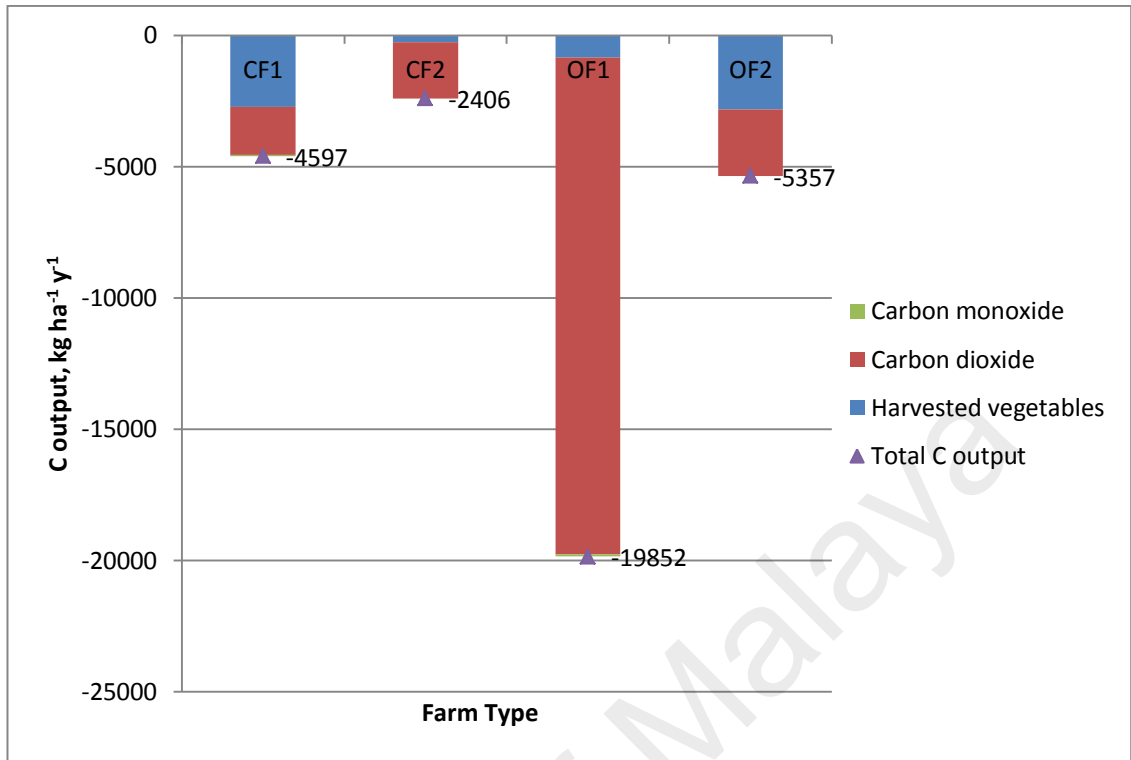


Figure 4.21: Annual cumulative output fluxes of C, kg ha<sup>-1</sup> year<sup>-1</sup>

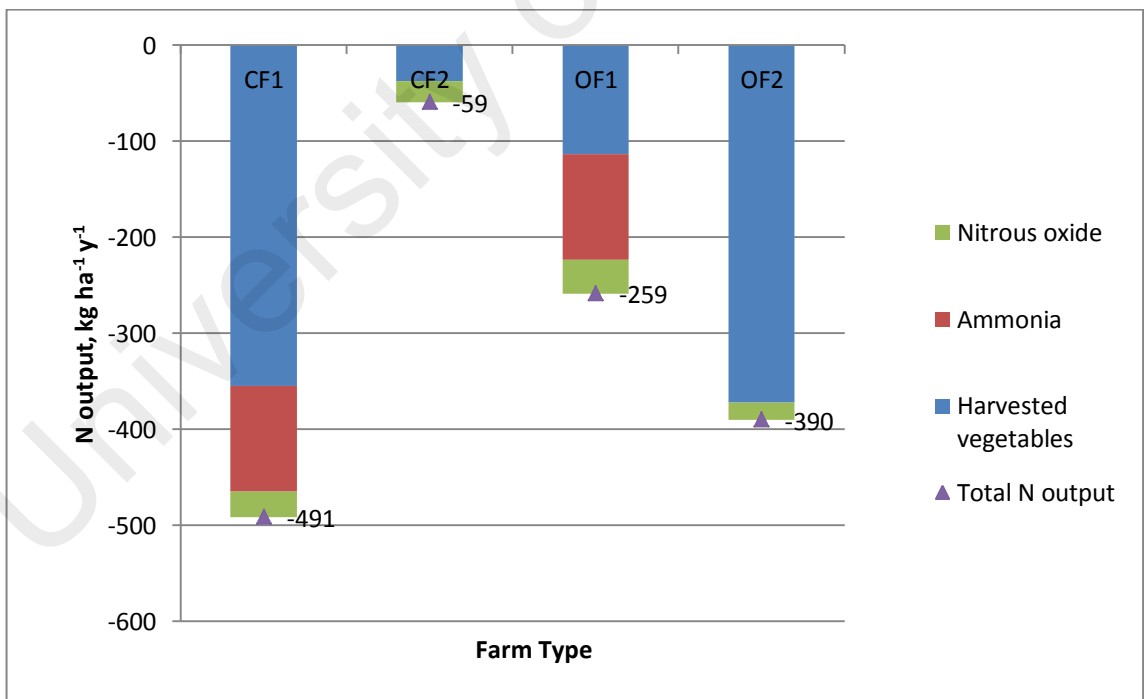


Figure 4.22: Annual cumulative output fluxes of N, kg ha<sup>-1</sup> year<sup>-1</sup>

## 4.6 Soil C and N content

The soil C and N content over the study period was analysed for soils in 10cm and 30cm depth. The monitoring of soil C and N concentrations were able to reliably describe the soil C and N changes (Farmer *et al.*, 2014; Lee *et al.*, 2009). The soil C and N stocks were incorporated into the STAN flow models for more comprehensive modelling. The soil C and N stocks data included under the process of “Farm land” which is where the soil stock is located. The soil C and N stocks were estimated based the soil C and N concentrations and soil bulk density (Figure 4.23).

### 4.6.1 Soil C content

The soil C concentrations (10 cm soil) in the farms in this study range from 1.30 % in CF1 to 5.47% in OF1, while for 30 cm soil depth the C concentrations range from 0.86 % in CF1 to 3.917 % in OF2. Comparable results were obtained from Li *et al.* (2015), where the mean soil C concentrations ranging from 1.82 to 3.45% between 1981 and 2011.

The average soil C concentrations for 10 cm soil in organic and conventional systems was  $3.50 \pm 1.05$  % and  $2.14 \pm 0.57$  % while for 30 cm soil were  $3.07 \pm 0.36$  % and  $1.26 \pm 0.261$ %, respectively. The average soil C content in organic system was significant higher than conventional system ( $P < 0.001$ ). This was supported by Gattinger *et al.* (2012) where the soil C concentration in organically managed soil was  $0.18 \pm 0.06$ % higher than the non-organical. During the study, the soil C concentrations in soil depth of 10 cm and 30 cm were increasing in OF1 and OF2 while it decreased in CF1 and CF2 (Figure 4.24 and 4.25).

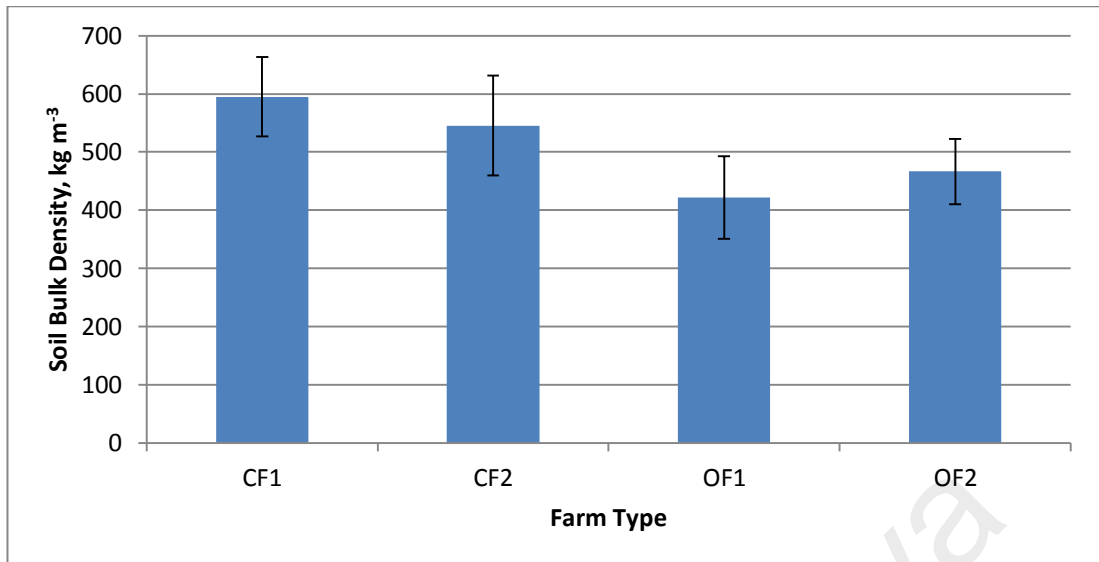


Figure 4.23: Soil bulk density for top soil in farms in this study

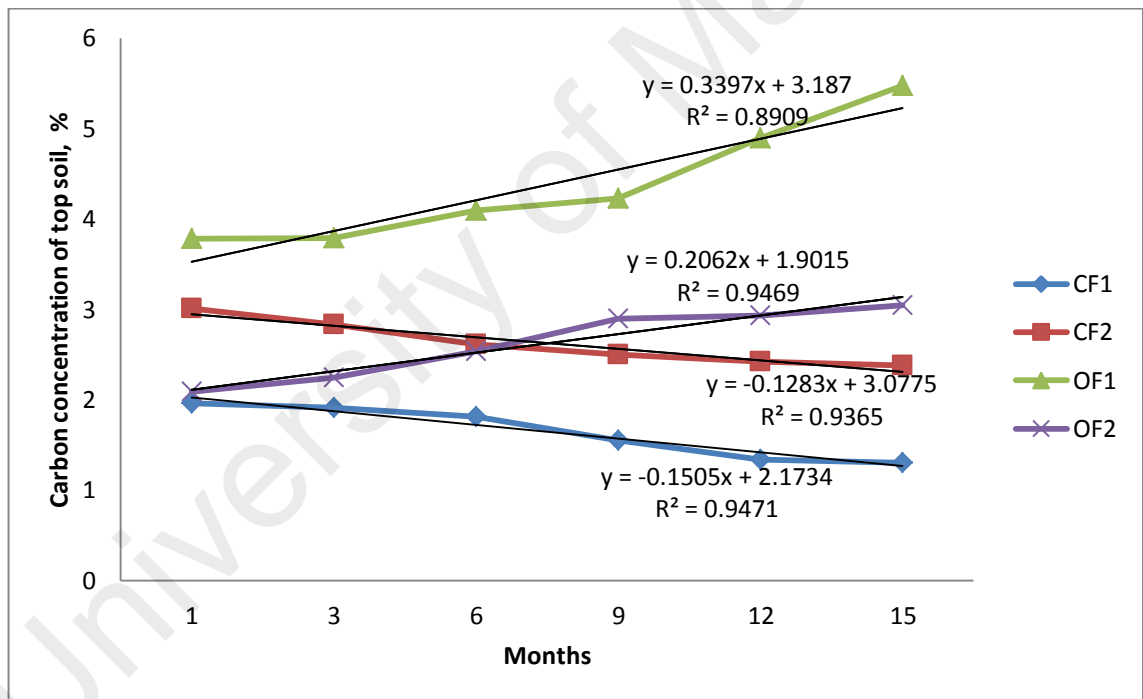


Figure 4.24: C concentrations trend in top soil (10cm)

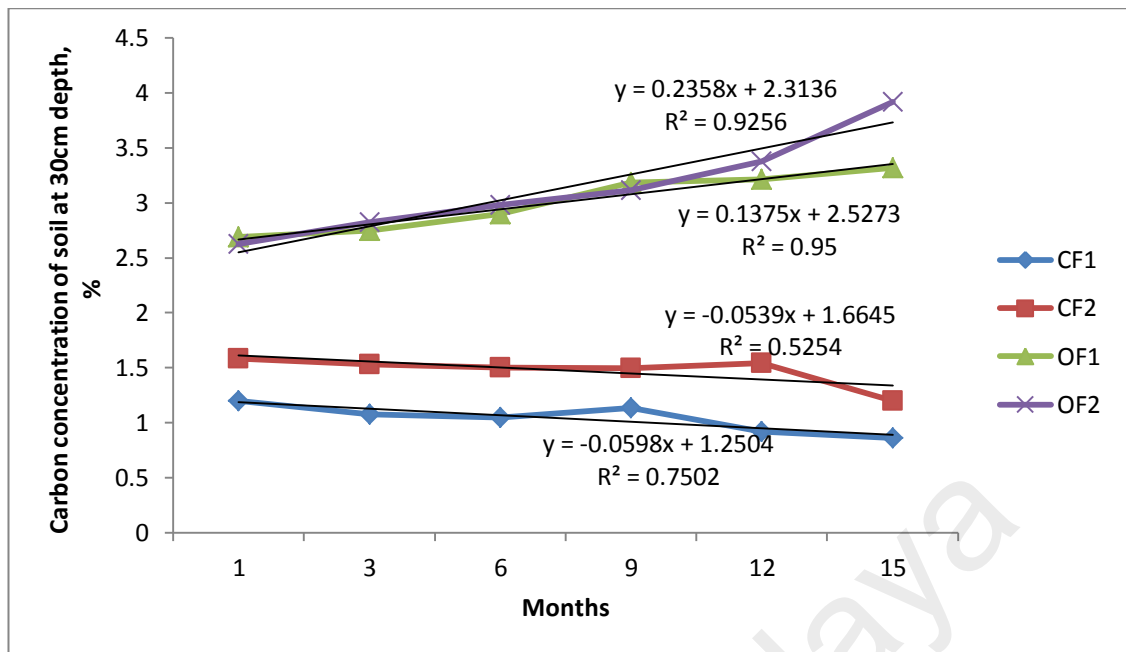


Figure 4.25: C concentrations trend in soil at 30cm depth

The changes were distinctively different between soil depth and also farm systems. During the study, the soil C concentrations in 10 cm soil increased at  $33.8 \pm 28.1\%$  and  $19.2 \pm 12.9\%$  in OF1 and OF2. Similar trend were observed for 30 cm depth of soil which organic systems had an increase of soil C concentration at  $12.6 \pm 10.0\%$  and  $25.9 \pm 16.6\%$  in OF1 and OF2, respectively (Figure 4.26 and 4.27).

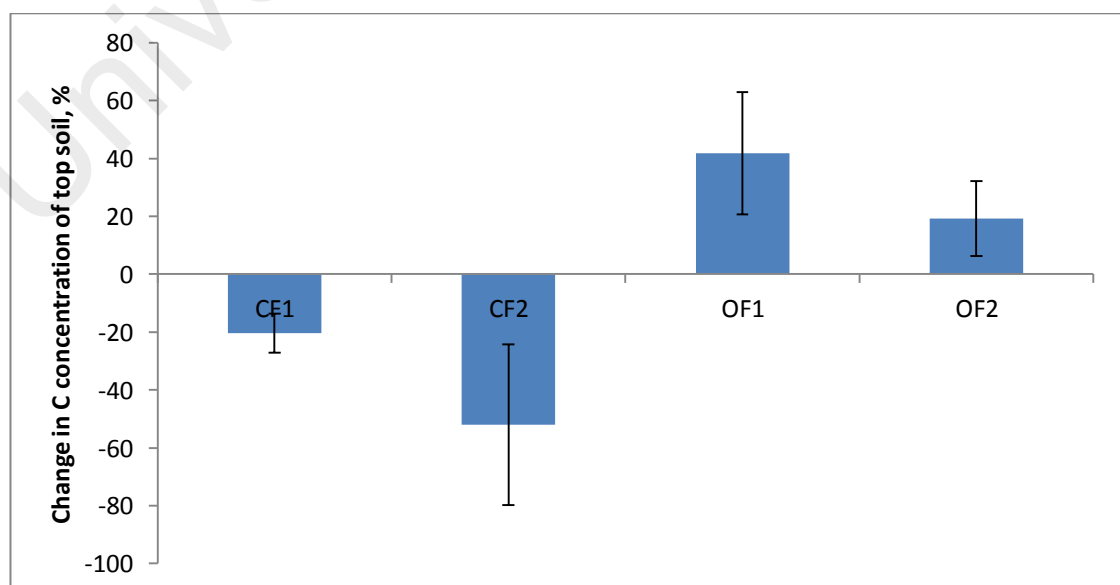


Figure 4.26: Change of C concentrations change in top soil (10cm)

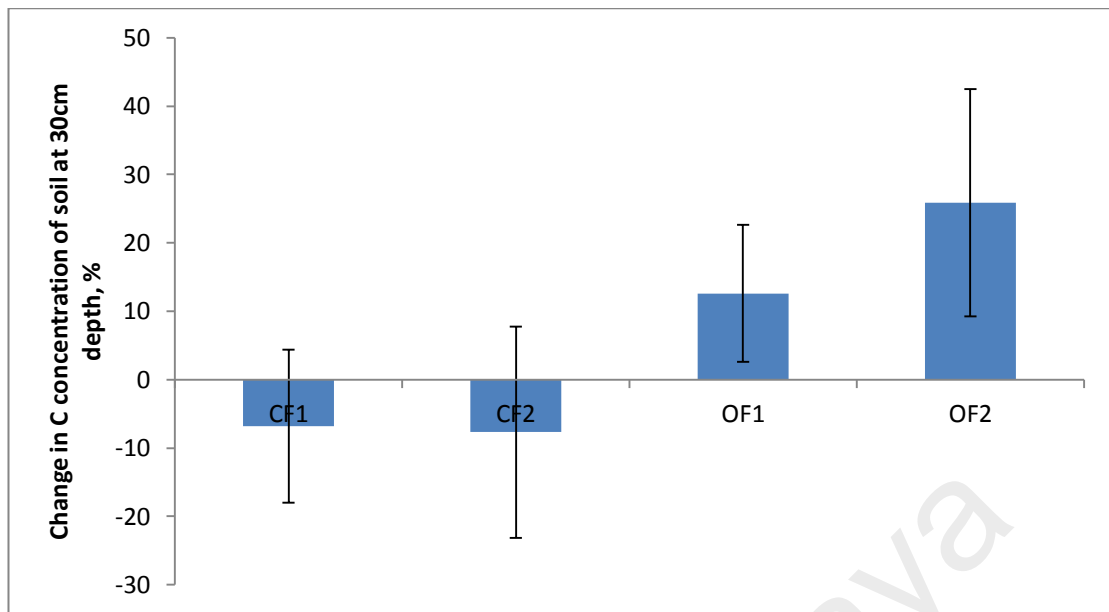


Figure 4.27: Change of C concentrations change in soil at 30cm depth

In CF1 and CF2 a reduction of  $13.1 \pm 10.1\%$  and  $12.6 \pm 7.21\%$  were observed in 10 cm soil depth. In 30 cm soil depth the soil C decreased by  $6.82 \pm 11.1\%$  and  $7.7 \pm 15.5\%$  in CF1 and CF2, respectively. The S.E value was higher than the mean value indicated the high uncertainty and variation of soil C concentrations in 30 cm soil depth during the study period. Similar to soil C concentration, the estimate of C stock was in the increasing trend in OF1 and OF2 but decreased in CF1 and CF2 (Figure 4.28) during the study period. In the first month of study the soil C stock in CF1, CF2, OF1 and OF2 were recorded 11,675, 16,429, 15,966, and 9,734 kg ha<sup>-1</sup>, respectively. At the end of the study the soil C in CF1 and CF2 decreased by 3,132 and 2,749 kg ha<sup>-1</sup> y<sup>-1</sup> while in OF1 and OF2 it increased by 5,706 and 3,587 kg ha<sup>-1</sup> y<sup>-1</sup>, respectively. The results of OF1 were similar with a long-term experiment that recorded cumulative sequestration of 7,400 kg C ha<sup>-1</sup> y<sup>-1</sup> for various cropping and tillage systems (Barbera *et al.*, 2012). A meta-analysis indicated the soil C sequestration can range from -3,380 to 3,083 7,400 kg C ha<sup>-1</sup> y<sup>-1</sup> (Puget & Lal, 2005). In general, the average C stock in organic systems ( $15,355 \pm 3,976$  kg ha<sup>-1</sup>) were significantly (P=0.028) higher than conventional systems ( $12,067 \pm 2,792$  kg ha<sup>-1</sup>). Similar results were reported by Clark *et al.* (1998)



where the soil C in 0-15cm depth in organic farms were significantly higher than in conventional farms. However, no significant differences were observed in soil depth of 15-30cm. This was supported by Gattinger et al. (2012) who reported the soil C stocks in organically managed soil were  $3,500 \pm 1,080 \text{ kg C ha}^{-1}$  higher than non-organically managed soil. Different results were reported by a long term study in Sweden which concluded the lack of evidence of C sequestration in both organic and conventional farms (Lynch *et al.*, 2012). The author also reported decreased of soil organic C concentrations in both farming systems. However, the magnitude of decrease was lesser in organic systems due to the high C input and low soil pH. The study by Parras-Alcántara & Lozano-García (2014) also indicated no significant differences between organic and conventional farm in C sequestration and the authors further concluded that management practices have little influences in soil C sequestration. Different results observed were mainly due to the complex mechanisms that governed the soil C flux such as crop type (rhizo-deposits), soil quality (bulk density, C content, porosity), soil management history, climate and landscape (Govaerts *et al.*, 2009).

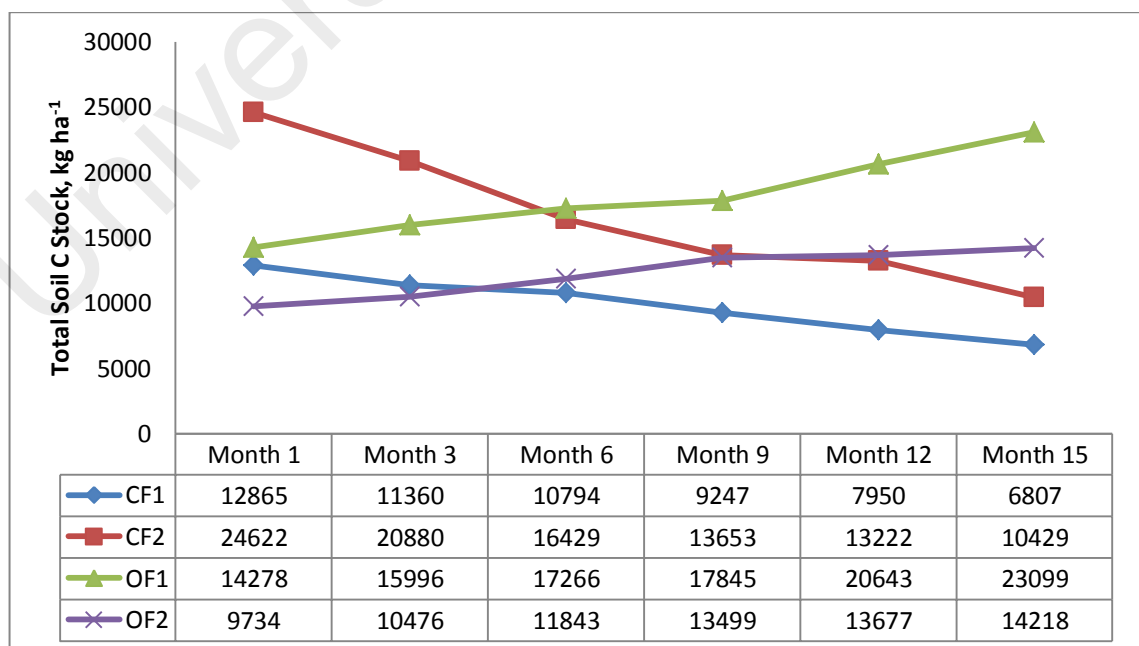


Figure 4.28: Soil C stock

The higher rate of compost application in OF1 ( $206 \text{ t ha}^{-1} \text{ y}^{-1} \approx 18,680 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) might be the reason for the increase of soil C as compared to OF2 ( $34 \text{ t ha}^{-1} \text{ y}^{-1} \approx 11,871 \text{ kg C ha}^{-1}$ ). A study on the effect of compost and scrap mixture towards SOM indicated compost C:N ratio of 1:15 and 25:1 can effectively and long lastingly increase 60% and 55% of SOM (Scotti *et al.*, 2015b). The increased SOM can contribute to increased soil C which suggested 10%-20% of C from the applied organic matters were stored as C pool (Rosen & Allan, 2007). Thus, the compost application in organic systems may be the cause for increased soil C stock.

Both CF1 and CF2 applied chicken manure into the soil. However, the grower of CF1 also included compost into the nutrient regime. A study demonstrated that short term application of compost and manure increased C level in top soil (0-25cm) by 41% and 25% while soil without organic matters input decreased by 3% (Fronning *et al.*, 2008). The results suggested higher contribution of SOM from compost application as compared to soil amended with manure. Thus, the lack of compost input in CF2 might be the reason for higher soil C reduction than CF1. However, Leifeld *et al.* (2009) highlighted the manure application in conventional farm resulted in no significant difference in soil C sequestration as compared with organic farms. Some suggested lower soil disturbance such as conservation tillage is the major factor for increasing soil C (Six *et al.*, 1999). However, in this study the tillage practices have little influence over soil C as both organic and conventional farm have similar tillage system. In addition, there were studies that found no significant C sequestration with conservation tillage due to complex mechanisms and pathways of soil C and highly variable magnitude effect of tillage practice (Puget & Lal, 2005).

The soil C:N in the farms in this study ranged between 7:1 and 15:1 which was below the threshold ratio that limit the microbial growth. Study showed that the the ideal C:N ration for long term C storage should be above 25:1-30:1 (Rosen & Allan, 2007). Even though OF1 and OF2 were identified as potential C sequester but with the low soil C:N the C storage is highly vulnerable towards microbial decomposition. Therefore, several authors highlighted the influences of C:N, history of the soil, soil quality, soil management, crop types, and climate towards soil C stock changes (Govaerts *et al.*, 2009; Li *et al.*, 2015; Neill *et al.*, 1997; Parras-Alcántara & Lozano-García, 2014).

#### 4.6.2 Soil N content

The soil N concentrations at 10 cm soil and 30cm soil in CF1 and CF2 decreased steadily during the study period (Figure 4.29 and 4.30).

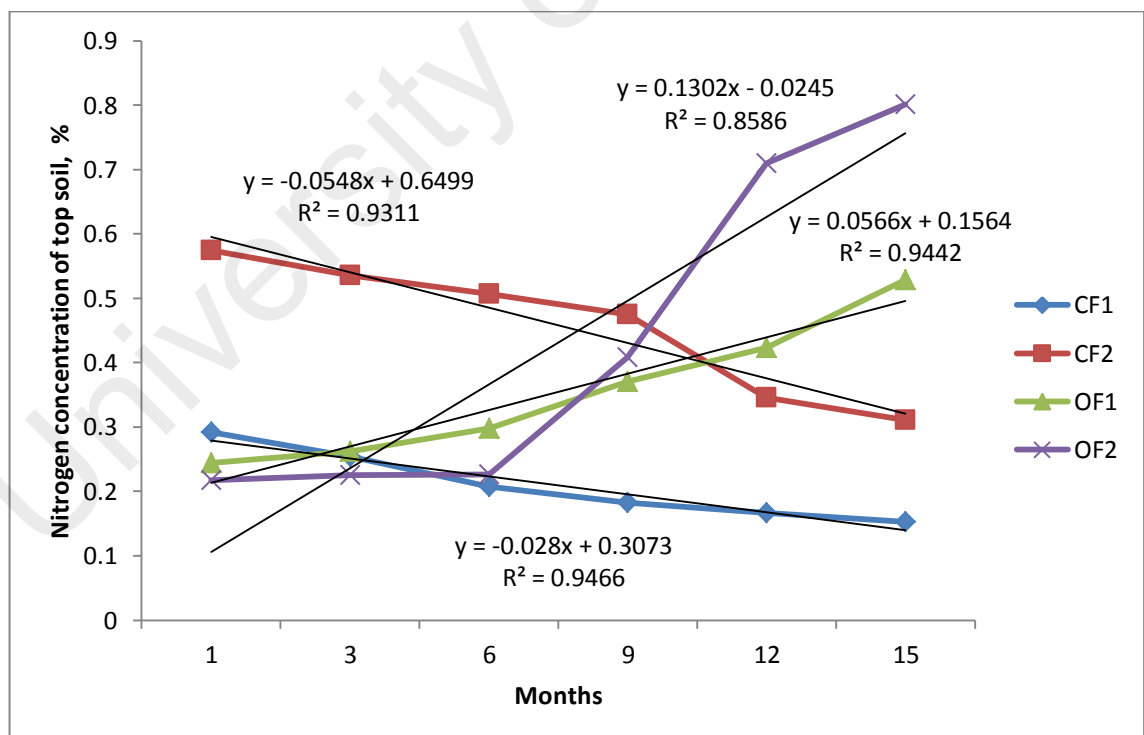


Figure 4.29: N concentrations trend in top soil (10 cm)

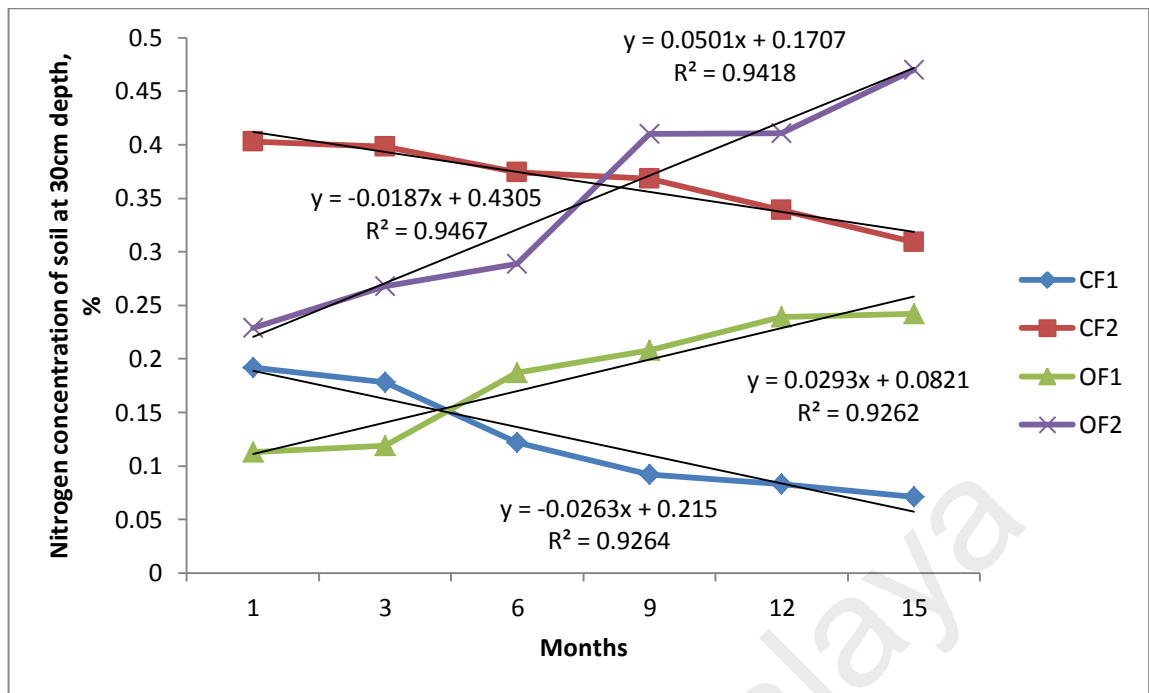


Figure 4.30: N concentration trend in soil at 30 cm depth

At 10 cm depth of the farms soil in this study, the soil N concentrations ranges from 0.153% in CF1 to 0.801% in OF2, while for 30 cm soil depth it ranges from 0.071 % in CF1 to 0.470 % in OF2. The average soil N concentrations at 10 cm depth of soil from organic and conventional system was  $0.393 \pm 0.196\%$  and  $0.334 \pm 0.023\%$ , respectively, while at 30 cm depth, it was  $0.265 \pm 0.113\%$  and  $0.244 \pm 0.133\%$ , respectively. Similar results were reported by Puget & Lal (2005) where the three farm systems: no till, chisel and plow have soil N that ranged from 0.2-0.3% in 10 cm soil and 0.1-0.2 % in 30 cm soil. This was supported by Bowles et al. (2014) who reported the average soil N of 13 organic farms was 0.15% (range: 0.08-0.210%). Lower soil N concentration was observed in pasture areas which was about 0.1-0.15% in 0-10 cm soil and 0.05-0.1% in 30 cm soil (Groppo *et al.*, 2015). Higher soil N in this study might be due to higher total N input in the farm systems and variation in soil properties. Similar to Groppo et al. (2015) the soil N in this study decreased with soil depth whereby the soil N in 30 cm soil depth was lower than that at 10 cm soil depth. The average soil N in organic system

was slight higher than conventional system ( $P < 0.05$ ) and similar results were observed in Clark et al. (1998) who recorded that the soil N concentrations (0-30 cm) in organic farms were  $0.12 \text{ g kg}^{-1}$  higher than conventional farms.

In CF1 and CF2, the average reduction of  $2.78 \pm 1.43 \%$  and  $5.26 \pm 4.28\%$  were observed in 10 cm soil depth while in 30 cm soil depth the soil N decreased by  $2.42 \pm 1.95 \%$  and  $1.88 \pm 1.23\%$ , respectively (Figures 4.31 and 4.32). The soil N reduction in CF1 and CF2 might be due to higher N loss via runoff, leaching, or volatilization than total N input in the farm systems (Cade-Menun *et al.*, 2013; Ju *et al.*, 2011; Stenberg *et al.*, 2012; van Eerd & Fong, 1998; Wang *et al.*, 2014).

The N concentrations in OF1 and OF2 for 10 cm and 30 cm soil depth gradually increased during the study period. The average soil N concentrations in 10 cm soil depth increased at  $5.70 \pm 3.41\%$  and  $11.7 \pm 12.7\%$  in OF1 and OF2, respectively. Similarly, the 30 cm soil in OF1 and OF2 also had an increase of soil N concentrations at  $2.58 \pm 2.62\%$  and  $4.82 \pm 4.60\%$ , respectively. The soil C and N have both increased in organic systems and decreased in conventional systems suggested correlation between soil C and N. This was supported by Kiba et al. (2012) which showed that the total soil N was strongly correlated to the total soil C content and the study also indicated that the repeated organic matters application increased soil N content. The soil N increased in organic systems might be due to the repeated application of compost which significantly increased the soil N content (D'Hose *et al.*, 2014). Several authors also reported increased soil N through compost application (Chalhoub *et al.*, 2013; Doan *et al.*, 2015; Polo *et al.*, 2015). Study by Pimentel et al. (2005) highlighted 47%, 38%, and 17% of the N derived from organic animal, legume and synthetic fertilizer retained in the soil a year after application (Pimentel *et al.*, 2005). Further, a research showed that a single

application of yard waste compost increased soil N by 11.7% ( $770 \text{ kg ha}^{-1}$ ) after 10 years of application (Yang *et al.*, 2014b). However, the impact of compost application to soil C and N highly depends on compost type and application frequency. According to Yang *et al.* (2014) application rate of  $300 \text{ t ha}^{-1}$  or compost derived from yard waste or straw have greater and long lasting effects than the application of  $75 \text{ t ha}^{-1}$  or compost derived from food waste. Therefore, the variations of soil N concentrations in the farms in this study might be due to the amount and types of soil amendments applications.

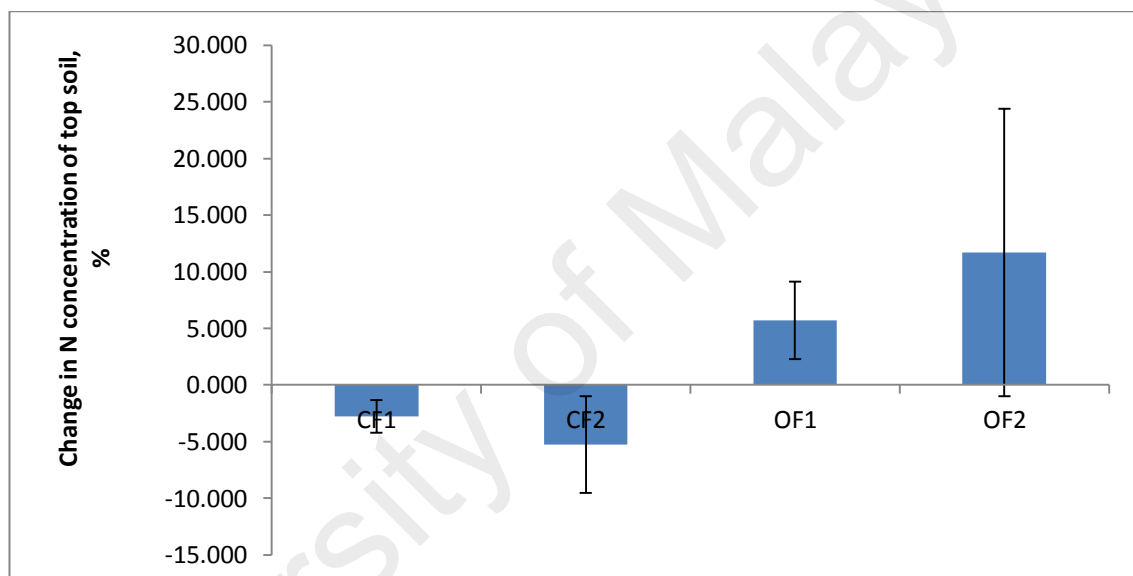


Figure 4.31: Change of N concentrations change in top soil (10 cm)

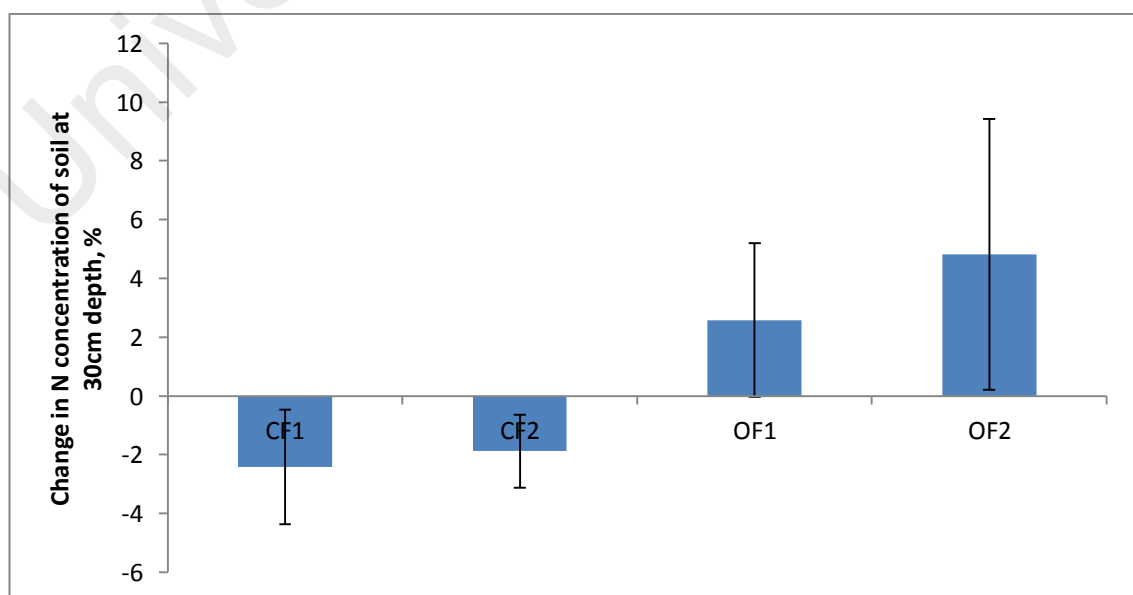


Figure 4.32: Change of N concentrations change in soil at 30 cm depth

The soil N stocks were in the increasing trend in OF1 and OF2 while the soil N stocks were reduced in CF1 and CF2 (Figure 4.33). In the first month of study the soil N stock in CF1, CF2, OF1 and OF2 were recorded as 1,738, 3,131, 1,030, and 1,013 kg ha<sup>-1</sup>, respectively. Similar soil N of 2270 and 6190 kg ha<sup>-1</sup> is observed in Bernard et al. (2012). The soil N stock in CF1 and OF2 were similar to a study on crop-livestock systems which the average soil N stocks was about 1,720 ± 729 kg ha<sup>-1</sup> (Groppo *et al.*, 2015). However, higher soil N stocks were observed in Puget & Lal (2005) which ranged between 5,000-7,000 kg ha<sup>-1</sup> for various cropping systems.

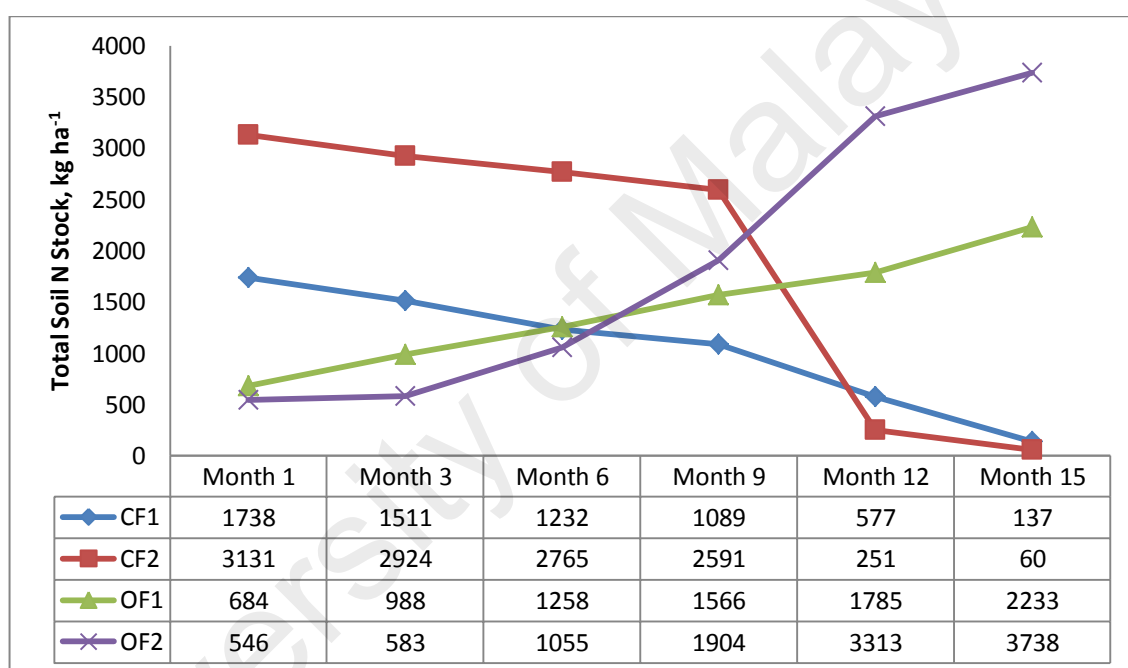


Figure 4.33: Soil N stock

At the end of the study the soil N stock in CF1 and CF2 decreased by 662 and 1,148 kg ha<sup>-1</sup> y<sup>-1</sup> and in OF1 while in OF2 it increased by 962 and 2,180 kg ha<sup>-1</sup> y<sup>-1</sup>. In general, no significant differences ( $P > 0.005$ ) were observed for the average N stock in organic systems (1,754 ± 923 kg ha<sup>-1</sup>) and conventional systems (1,872 ± 792 kg ha<sup>-1</sup>). A study also recorded a decreased of 14% soil N stock (440 kg N ha<sup>-1</sup>) in soil depth of 0-17.5 cm under conventional managed farms (Diekow *et al.*, 2005). The soil N in all the farm systems had over the recommended soil N of 50 kg N Ha<sup>-1</sup> for 0-90cm soil depth

(Zhang *et al.*, 2013). This suggested potential N loss from the farms in this study via volatilization, runoff or leaching that might impact the environment. In addition, the soil C:N in the farms in this study ranged between 7:1 and 15:1, thus it is likely the soil N is vulnerable for mineralization by soil microbes (Rosen & Allan, 2007).

The S.E value for soil N indicated high uncertainty and variation of soil N concentrations during the study period. The rates of change in soil N were distinctively different between soil depths and farm systems. This was supported by Bowles *et al.* (2014) and Bowles *et al.* (2015) which revealed the soil  $\text{NH}_4^+$  and  $\text{NO}_3^-$  pools were highly variable across fields, sampling times, and soil depths. In addition, the fertilization practices and site history also contributed to 96-97% of variability in soil chemical properties (Kiba *et al.*, 2012).

#### **4.7 Water management**

Water management differed between organic and conventional systems and also individual farms. However, one similarity observed in all the farms in this study was the establishment of catchment pond nearby the farms. The catchment pond collects rainfall and runoff from nearby farms and the water in the pond is used for irrigation. The water flow model was developed with the assumption that all water runoff were collected by the catchment pond near the farms. This is to evaluate the metabolism of C and N flow via water flux and to assess potential C and N stock in catchment pond. The physical-chemical water of rainfall, irrigation, runoff and leachate were analysed to understand the water quality and condition (Appendix C-E).



#### 4.7.1 Water flow model

Rainfall is the main source of water which enters the “Farm Land” directly as in CF1 and CF2 or indirectly for OF1 and OF2 via catchment pond. The water input rate in CF1 and CF2 varies from year to year because rainfall is the direct input of “Farm Land”. This is because climate induced differences and other uncertainties are involved in water quality status and trends assessment (Rozemeijer *et al.*, 2014). The yearly irrigation rate in CF1 and CF2 was  $5,316 \pm 1,620$  and  $9,920 \pm 1,542$  t ha<sup>-1</sup> y<sup>-1</sup> which is lower in comparison to OF1 ( $19,282 \pm 3,148$  t ha<sup>-1</sup> y<sup>-1</sup>) and OF2 ( $15,815 \pm 7,260$  t ha<sup>-1</sup> y<sup>-1</sup>). The lower irrigation rate in conventional systems are mainly because farmers in CF1 and CF2 irrigate the crop only on non-rainy days.

The artificial catchment ponds near the farms in this study were used to collect rainfall, store water and runoff from farms and to supply water for farms use. The water input in the catchment pond of conventional systems CF1 and CF2 was solely from rainfall which was about  $26,194 \pm 3,207$  and  $22,799 \pm 2,278$  t ha<sup>-1</sup> y<sup>-1</sup>, respectively (Figures 4.34 and 3.35). In contrast, the catchment pond in OF1 relied on rainfall and river water in order to supply enough water for farming activities. Yearly, about  $4,000 \pm 500$  t ha<sup>-1</sup> y<sup>-1</sup> of river water were channelled into the catchment pond. The small size of the catchment pond in OF1 was unable to collect enough rainfall for farm use thus channelled extra water from nearby rivers. Catchment pond was established in OF2 to collect rainfall for irrigation purposes. Unlike the other three farms, a systematic rainfall collection drains are installed in the rain shelter in OF2 which collect and channelled about  $20,440 \pm 2,044$  t ha<sup>-1</sup> y<sup>-1</sup> of rainfall into the catchment pond.

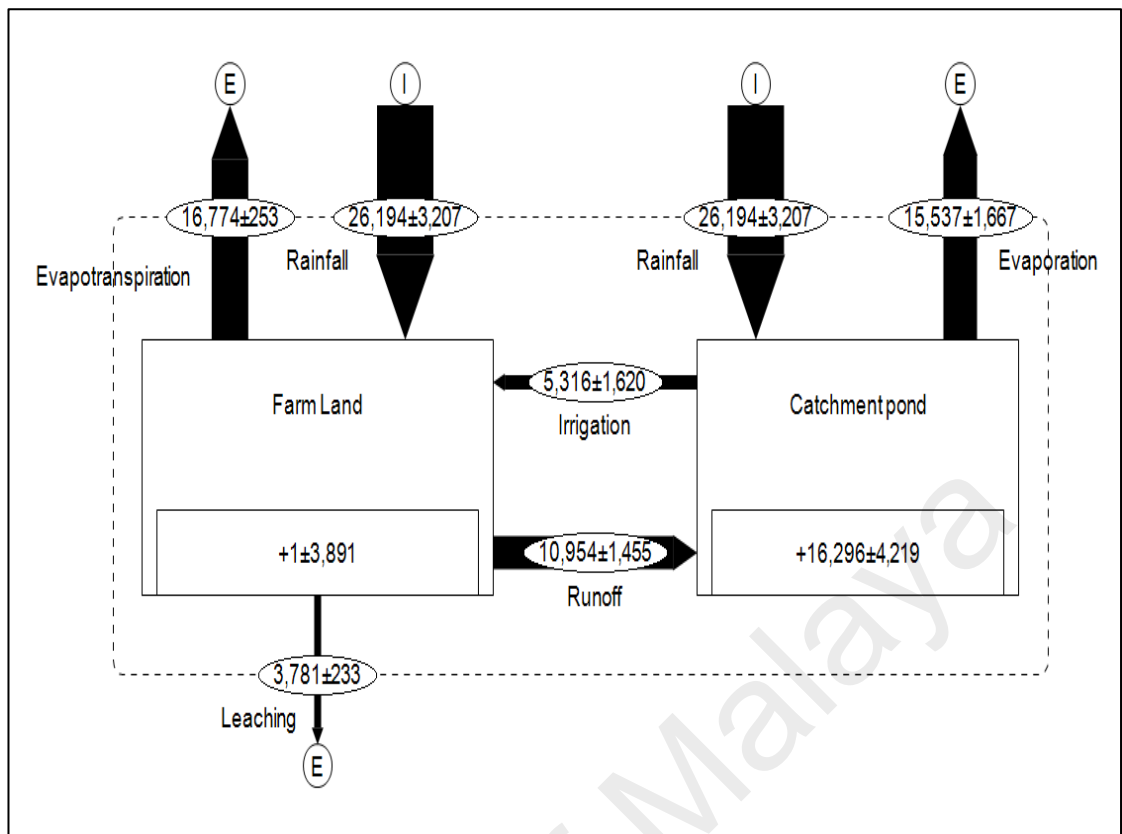


Figure 4.34: Water flow model for CF1 ( $t\ ha^{-1}\ y^{-1}$ )

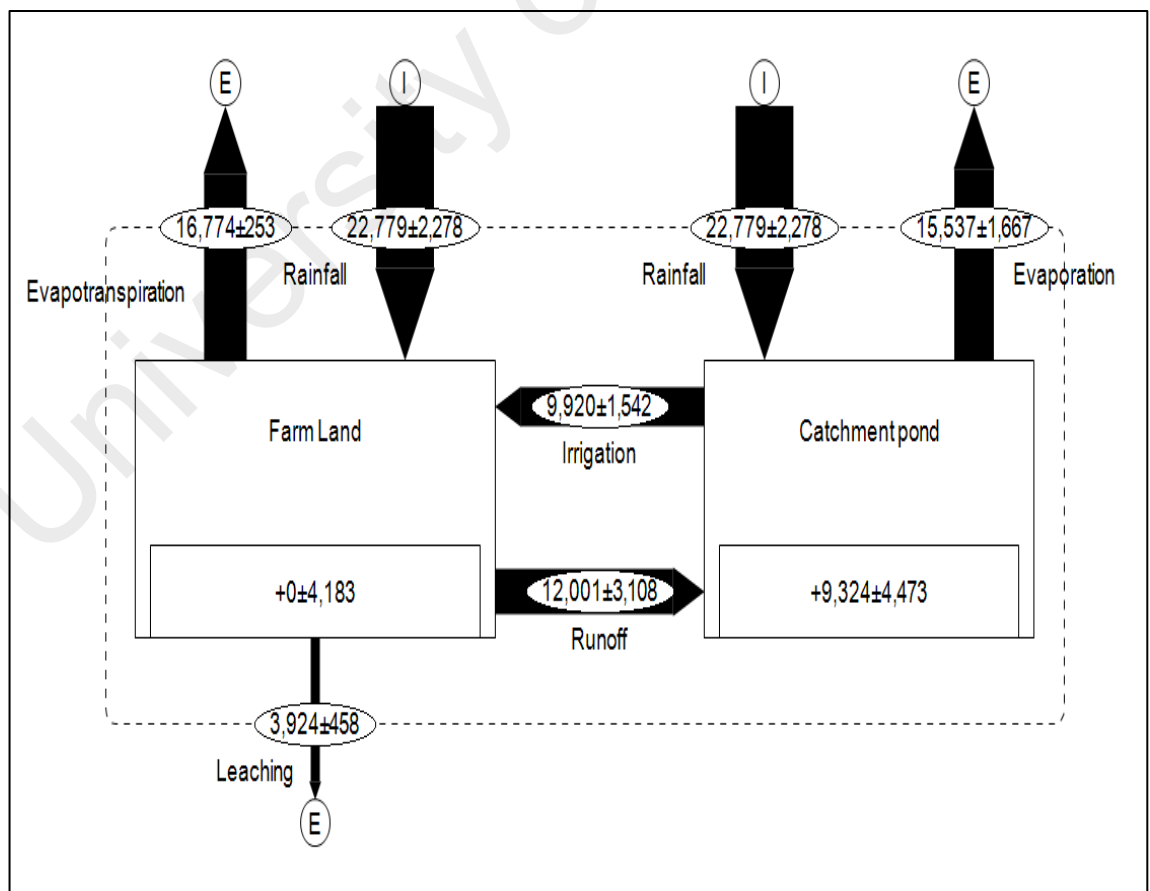


Figure 4.35: Water flow model for CF2 ( $t\ ha^{-1}\ y^{-1}$ )

The water flow model used in this study only estimates the total water output without indicating the volume of leaching (Bengtsson *et al.*, 2003). Thus, based on Song *et al.* (2009) the average leaching volume was assumed to be 0.12% of total irrigation. The estimated leaching from CF1, CF2, OF1 and OF2 was around  $3,781 \pm 233$ ,  $3,924 \pm 458$ ,  $2,314 \pm 378$  and  $1,898 \pm 871 \text{ t ha}^{-1} \text{ y}^{-1}$ , respectively (Figures 4.36 and 4.37). With the assumption that all runoff are collected in catchment pond, thus leaching and evapotranspiration were the only water output from “Farm Land”. The recycling of runoff from farm to catchment pond contributed water flow of  $10,954 \pm 1,455$ ,  $12,001 \pm 3,108$ ,  $8,407 \pm 2,641$  and  $5,355 \pm 6,259 \text{ t ha}^{-1} \text{ y}^{-1}$  in CF1, CF2, OF1 and OF2, respectively.

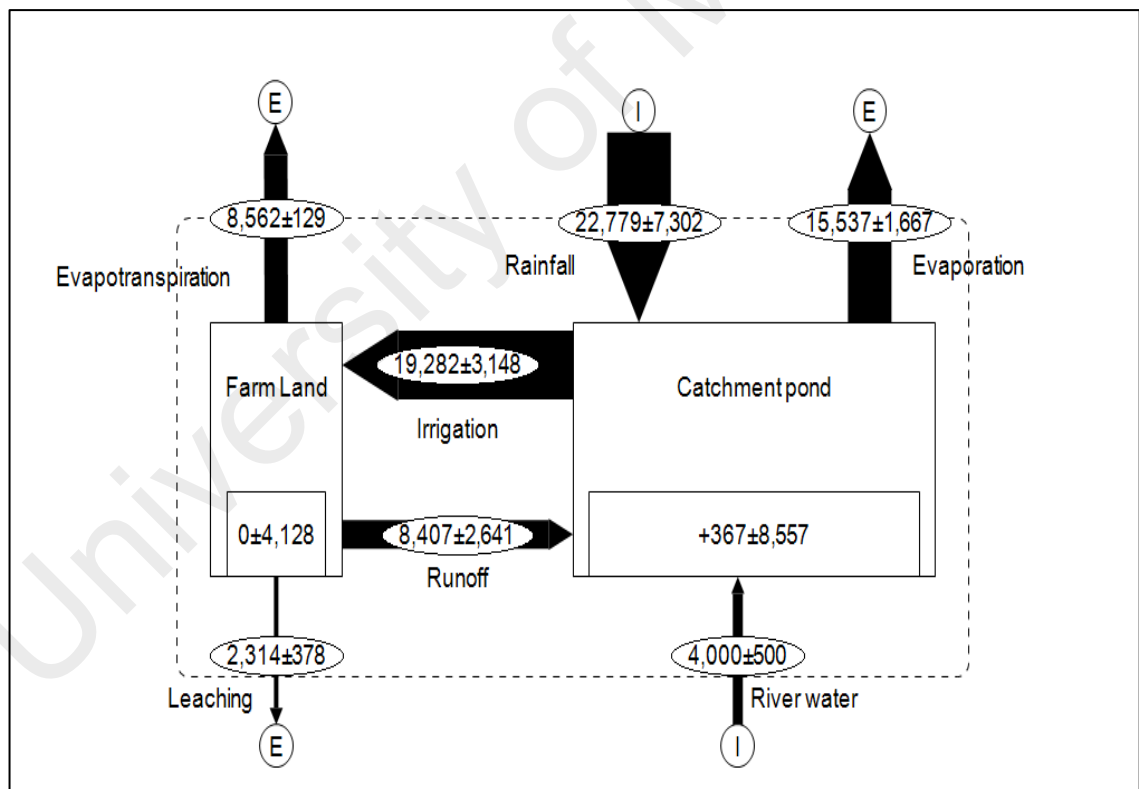


Figure 4.36: Water flow model for OF1 ( $\text{t ha}^{-1} \text{ y}^{-1}$ )

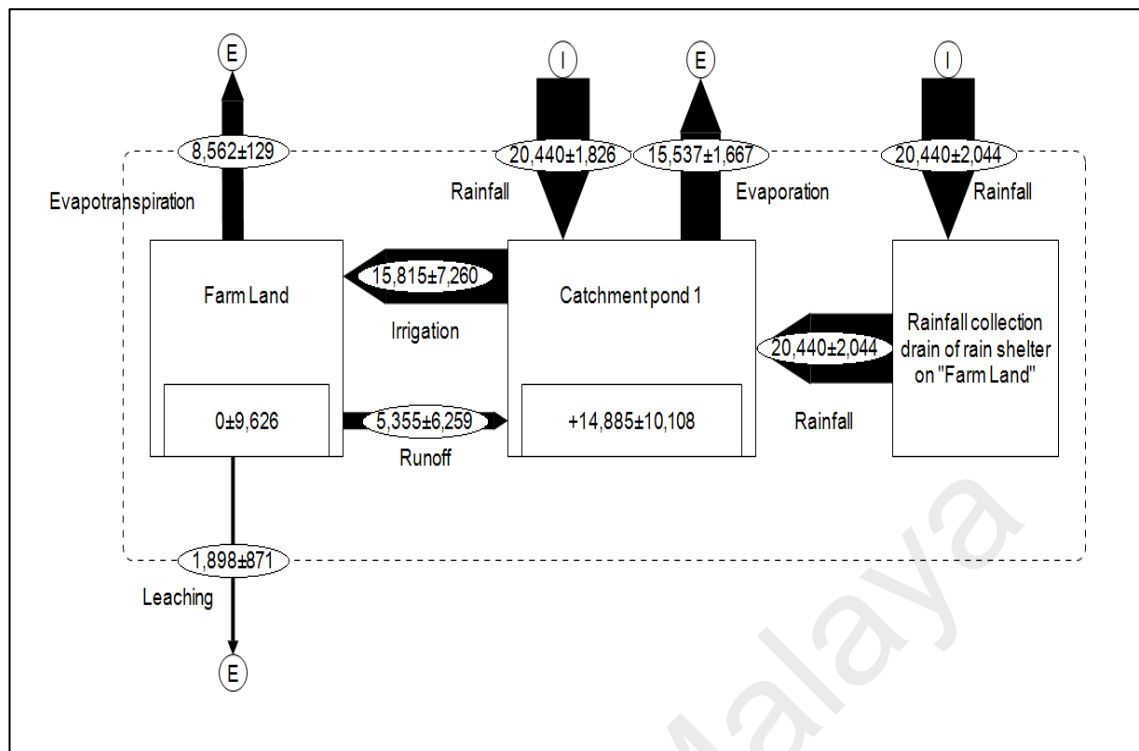


Figure 4.37: Water flow model for OF2 ( $t\ ha^{-1}\ y^{-1}$ )

#### 4.7.2 C flux via water route

Based on the water flow discussion in Section 4.7.1, the C flux via water route was established with the objective to understand C input-output via rainfall, irrigation, and leachate. Also, it was used to assess the potential C sequestration in catchment pond through runoff recycling from the farm systems. Rainfall and leaching were identified as the systems key input and output which transport C element into and out of the farms. Overall, the average C input via rainfall and irrigation contributed a total of 8.9, 134.1, 66.6 and 94.6  $kg\ C\ ha^{-1}\ y^{-1}$  into "Farm land" of CF1, CF2, OF1 and OF2, respectively (refer to section 4.3 for detail discussion). The irrigation activities had channelled  $3.1 \pm 0.5$ ,  $98 \pm 11.9$ ,  $66.6 \pm 3.9$ ,  $94.6 \pm 35.9\ kg\ C\ ha^{-1}\ y^{-1}$  of C from catchment pond into "Farm land" of CF1, CF2, OF1, and OF2, respectively (Figures 4.38 and 4.39).

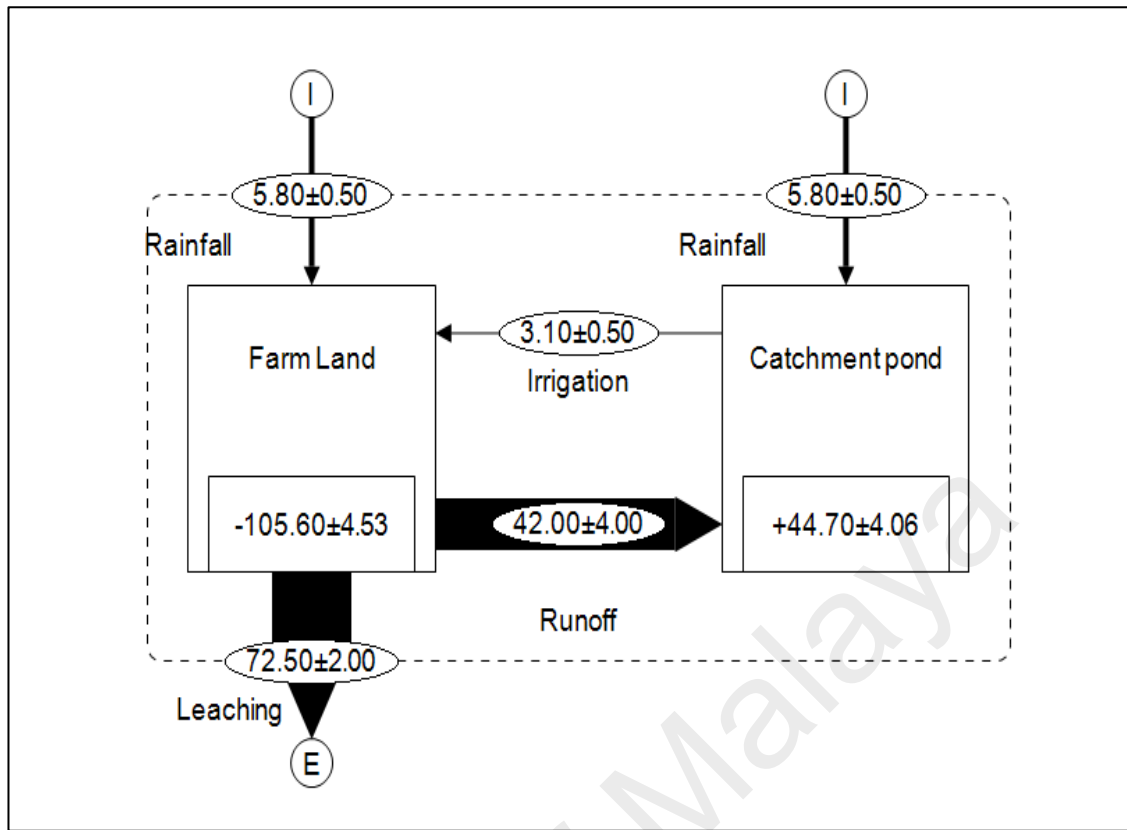


Figure 4.38: C flux through water route in CF1 (kg ha<sup>-1</sup> y<sup>-1</sup>)

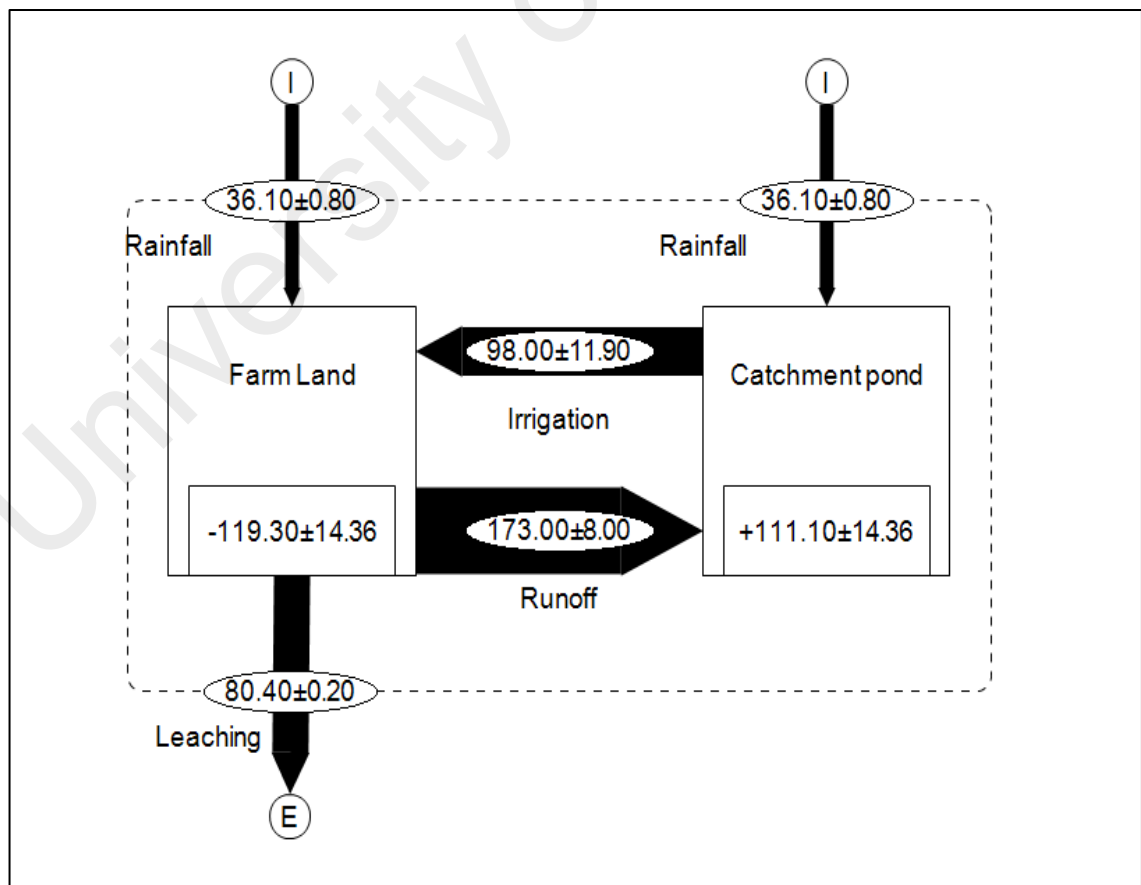


Figure 4.39: C flux through water route in CF2 (kg ha<sup>-1</sup> y<sup>-1</sup>)

The “Farm land” and “Catchment pond” were the two stocks identified in the water flow models which were able to store C in the systems. The “Farm land” in all the farms in this study experienced C deficit of  $105.6 \pm 4.53$ ,  $119.3 \pm 14.36$ ,  $327.50 \pm 44.27$  and  $14.80 \pm 45.70$  kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2, OF1, and OF2, respectively (Figures 4.40 and 4.41). The C lost from “Farm land” was mainly through runoff and leaching. The higher C output as compared to input indicated traces of soil C or organic matter being washed off from “Farm land” and may potentially cause soil degradation and depletion. Physical soil degradation often due to soil erosion by water force such as intensive irrigation and rainfall (Boulal *et al.*, 2011; O’geen & Schwankl, 2006; Oldeman, 1994; Snakin *et al.*, 1996). The total suspended solid (TSS) content in runoff in CF1, CF2, OF1 and OF2 were about  $0.157 \pm 0.119$ ,  $0.263 \pm 0.081$ ,  $0.250 \pm 0.107$  and  $0390 \pm 0.173$  mg L<sup>-1</sup>, respectively. This suggested high runoff erosion potential in OF2 and CF2 as compared to OF1 and CF1 as runoff TSS was commonly used as a preliminary indicator for runoff erosions (Bechmann *et al.*, 2008).

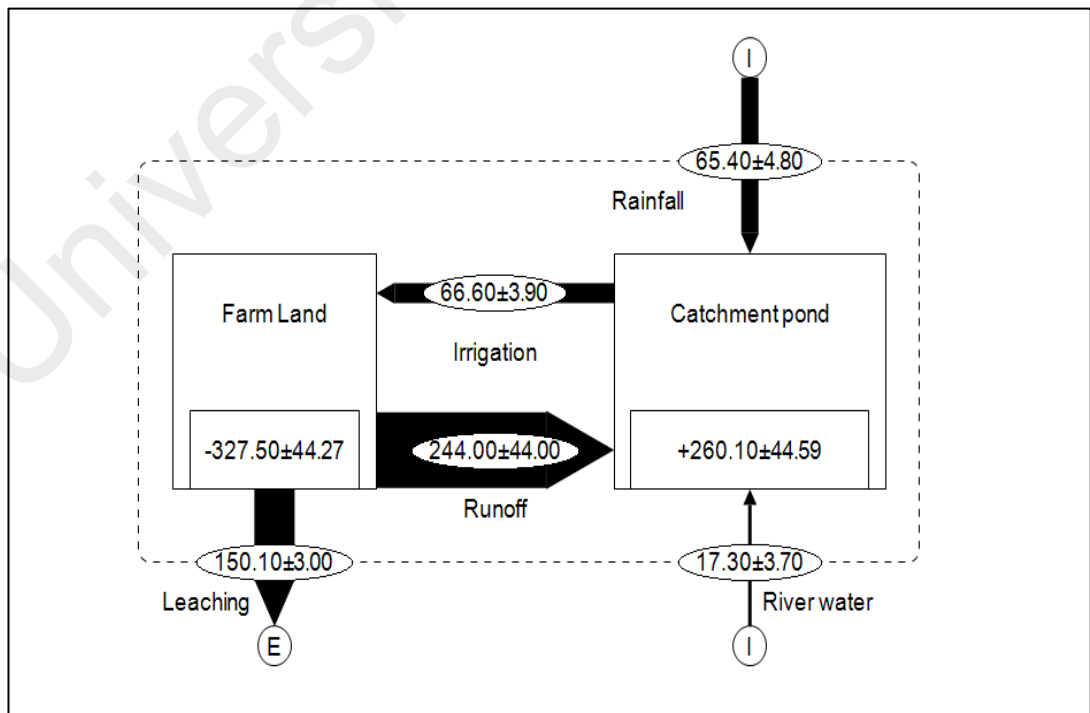


Figure 4.40: C flux through water route in OF1 (kg ha<sup>-1</sup> y<sup>-1</sup>)

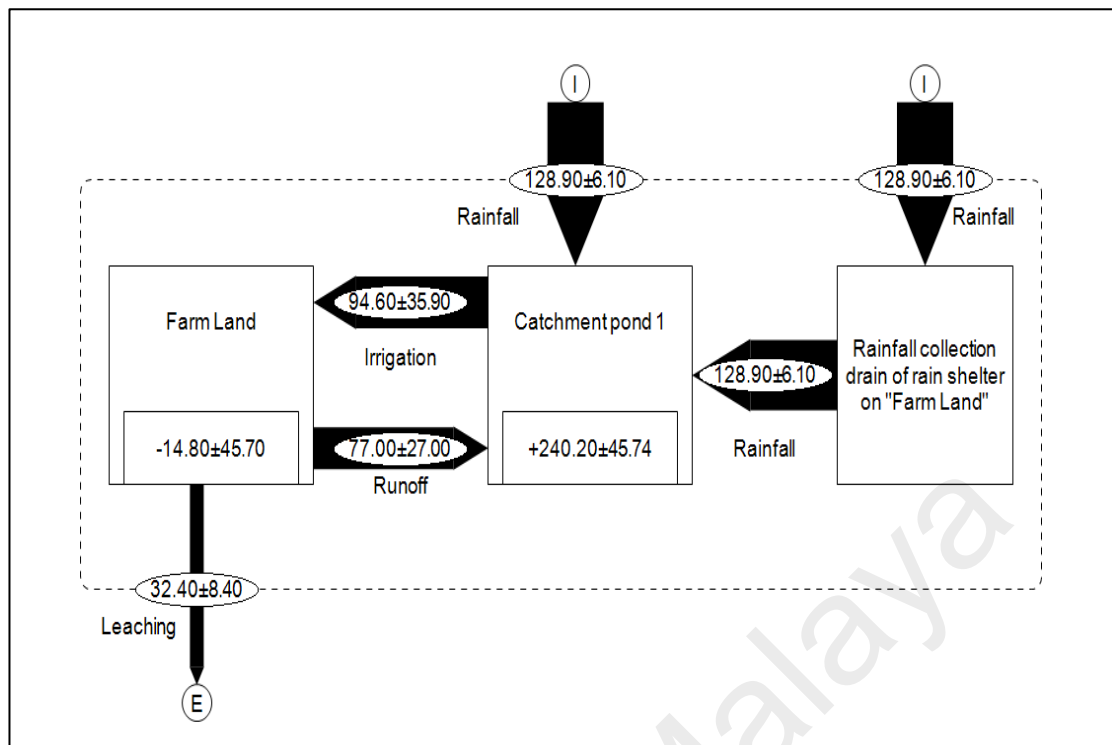


Figure 4.41: C flux through water route in OF2 (kg ha<sup>-1</sup> y<sup>-1</sup>)

During the study, average C runoff in CF1, CF2, OF1 and OF2 was about 3.866, 14.47, 29.11 and 14.42 mg L<sup>-1</sup>, respectively (Figure 4.42). Comparable results were reported by Van Gaelen et al. (2014) which the mean C concentration of runoff ranged between 4.7-10.1 mg L<sup>-1</sup>. The runoff C in OF1 was much higher as compared to other farms in this study and this might be due to the high compost application in the farm. Few studies indicated that the C content in runoff from soil amended with organic residues, compost and manure were higher than soil with and without mineral fertilizer application (Evanylo *et al.*, 2008; Van Gaelen *et al.*, 2014). Several authors indicated soil particle and nutrients from farm soils can easily be washed away by runoff (Pimentel *et al.*, 1995; Simonneaux *et al.*, 2015). Thus, recycling of runoff from farm was often considered a useful method to mitigate nutrient and pesticide impact on aquatic system (Newman *et al.*, 2014). According to water flow model, the recycling of runoff flow from "Farm land" to "Catchment pond" resulted in  $42 \pm 4.0$ ,  $173 \pm 8.0$ ,  $244 \pm 44$  and  $77 \pm 27$  kg C ha<sup>-1</sup> y<sup>-1</sup> of C flux (with the assumption that all runoff were

collected in “Catchment pond”). The C flux has contributed an average  $44.7 \pm 4.06$ ,  $111.1 \pm 14.36$ ,  $260.1 \pm 44.59$ , and  $240.2 \pm 45.74$  kg C ha<sup>-1</sup> y<sup>-1</sup> of C stock in “Catchment pond” of CF1, CF2, OF1, and OF2, respectively. Despite the fact that C is lost from “Farm land”, the runoff C that enters the “Catchment pond” was indirectly contributed to C sequestration. This is supported by Page *et al.* (2004) who reported soil erosion in pastoral stepland resulted as high as  $500 \pm 150$  kg C ha<sup>-1</sup> y<sup>-1</sup> entered the lake yearly and the lake was identified as C sink through C burial process (Nordt *et al.*, 2000). In addition, increased nutrient content in catchment pond due to runoff input may increase C burial rate due to increase of microbial biomass production rate due to eutrophication (Clow *et al.*, 2015; Dong *et al.*, 2012; Heathcote & Downing, 2012).

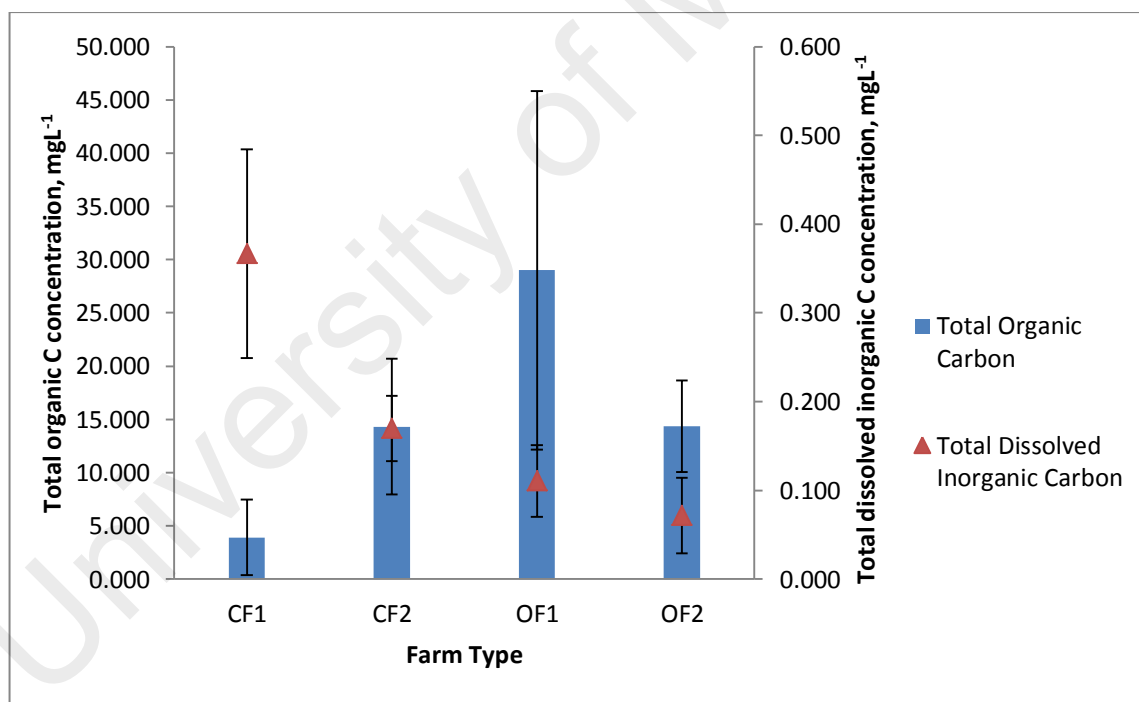


Figure 4.42: TOC and inorganic C concentrations in runoff

Leaching is the main pathway for C loss in the water flux model which resulted in  $72.5 \pm 2$ ,  $80.4 \pm 0.2$ ,  $150.1 \pm 3.0$ , and  $32.4 \pm 8.4$  kg C ha<sup>-1</sup> y<sup>-1</sup> in CF1, CF2, OF1 and OF2 respectively. The leachate C concentrations of  $65.1$  mg L<sup>-1</sup> was observed in OF1 which



was much higher in comparison to CF1 (19.2 mg L<sup>-1</sup>), CF2 (20.6 mg L<sup>-1</sup>), and OF2 (17.1 mg L<sup>-1</sup>) (Figure 4.43). Leachate C content similar to OF1 was reported by Nest et al. (2014) which the average total dissolved C in leachate samples was 74 mg L<sup>-1</sup>. The water analysis indicated that total organic C is the main component in rainfall, irrigation, runoff and leachate. Total organic C was mainly derived from organic matters such as manure and compost. Thus, the high organic C concentration in leachate and runoff of OF1 might be due to high composts application. In addition, the irrigation and rainfall intensity might also affect the leaching intensity which result in different C leaching in each farm system (Gao *et al.*, 2014). There is a possibility that the C that leached from the “Farm land” enters underground water or surface water which contributed to C sequestration. However, increased of C in water resource has detrimental effect on the ecosystem (Tuvendal & Elmqvist, 2011).

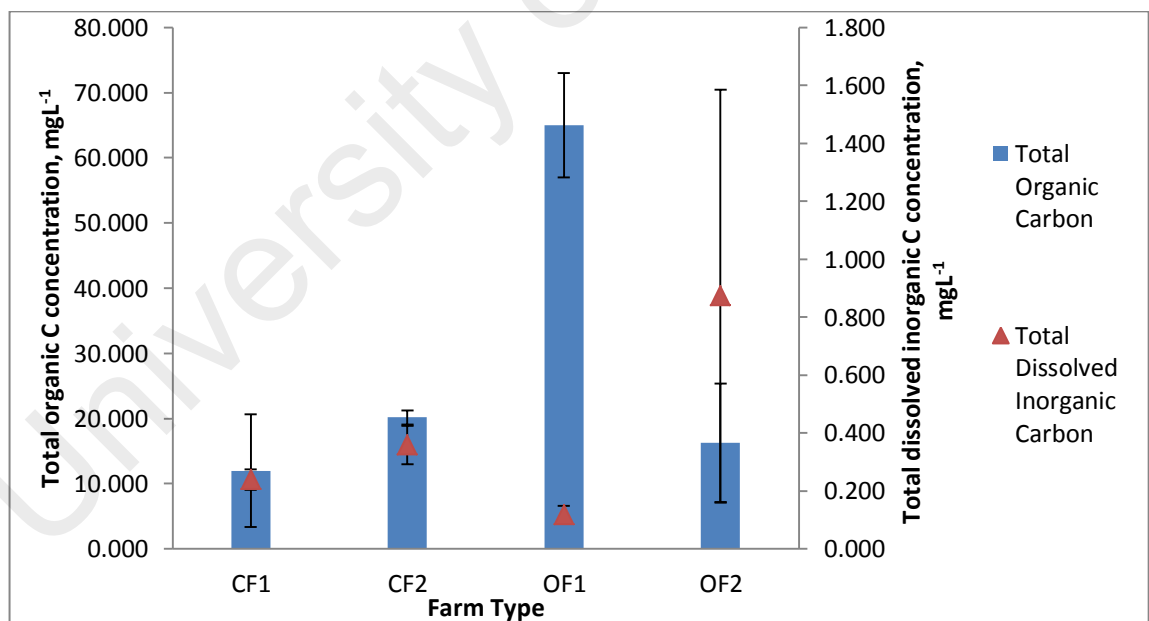


Figure 4.43: TOC and inorganic C concentrations in leachate

### 4.7.3 N flux via water input-output

Similar to C flux, the N water flux was generated based on the water flow model in Section 4.7.1. The N flow models and nutrient surplus models (Section 4.4 and 4.5) indicated N surplus in both organic and conventional farms. Thus, this section assessed the potential of N loss and stocks in the study farms. In addition, to quantify the N mass recycling from “Catchment pond” back to “Farm land” via irrigations. Rainfall and irrigation have contributed an average N input of 109.3, 128, 406.2 and 379 kg N ha<sup>-1</sup> y<sup>-1</sup> into “Farm land” in CF1, CF2, OF1 and OF2, respectively (Figures 4.44 and 4.45). A total of 50.9 ± 9.2, 102.3 ± 7.2, 406.2 ± 5.5 and 379 ± 110.6 kg N ha<sup>-1</sup> y<sup>-1</sup> flow via irrigation from “Catchment pond” to “Farm land” in CF1, CF2, OF1 and OF2, respectively (refer to Section 4.4 for details discussion on irrigation N and rainfall N).

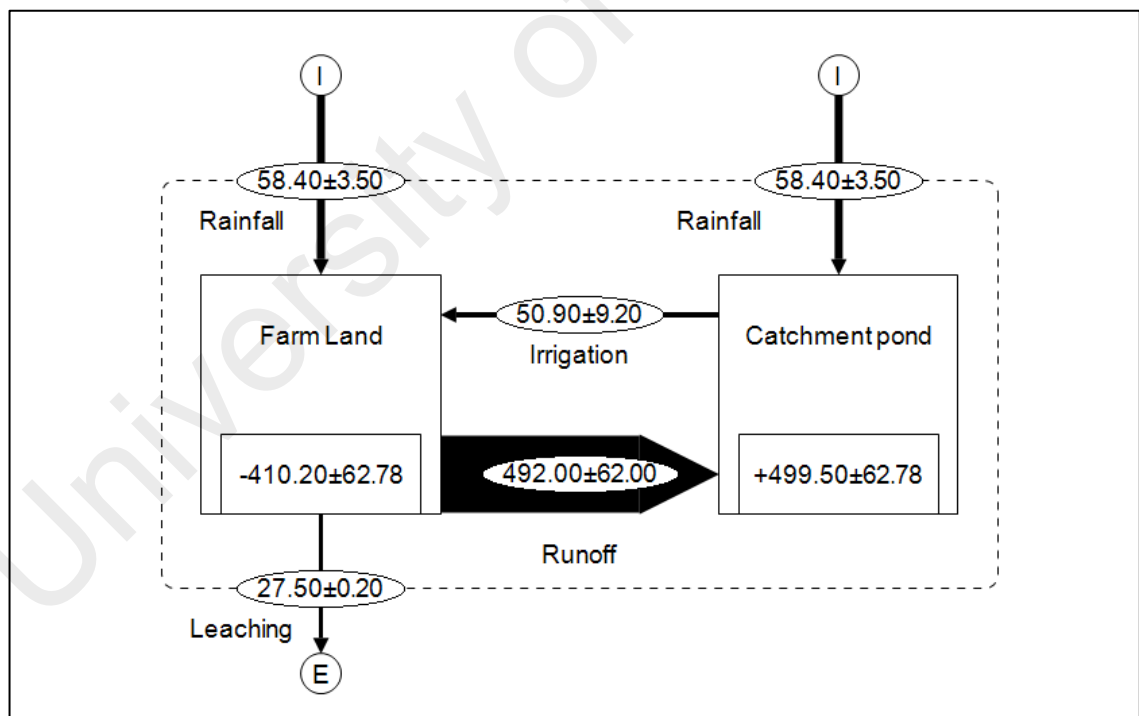


Figure 4.44: N flux through water route in CF1 (kg N ha<sup>-1</sup> y<sup>-1</sup>)

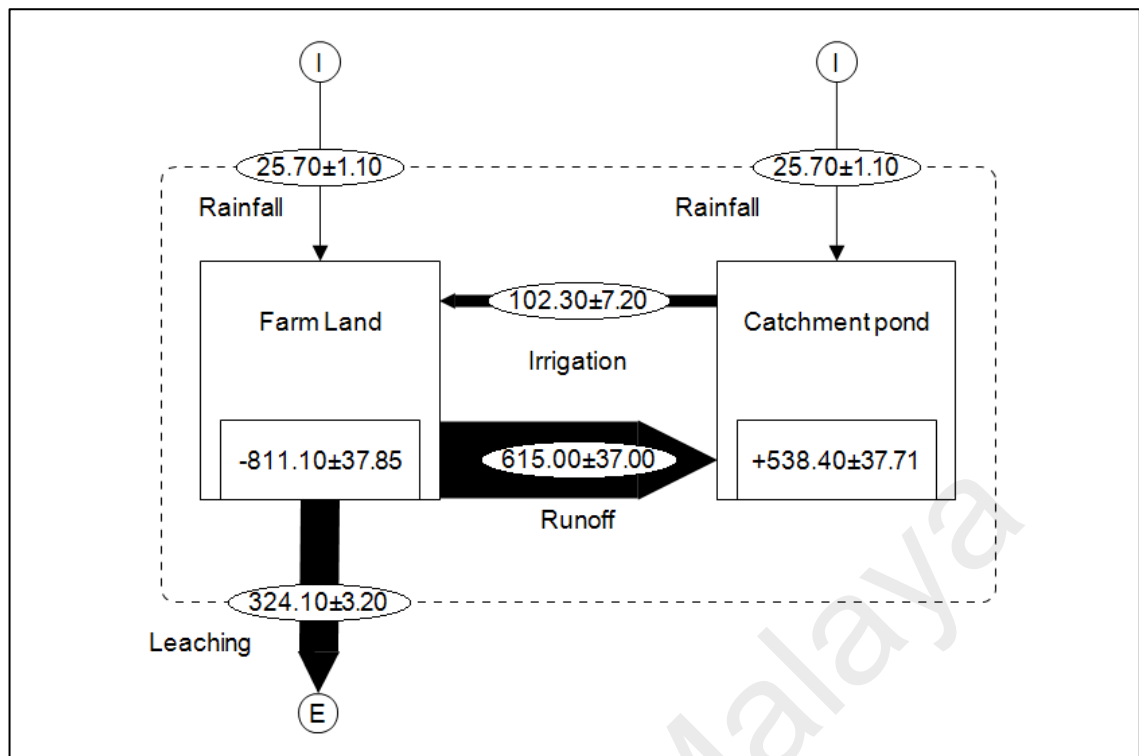


Figure 4.45: N flux through water route in CF2 ( $\text{kg N ha}^{-1} \text{y}^{-1}$ )

The N loss through runoff and leaching led to N stock deficit of  $410.2 \pm 62.78$ ,  $811.1 \pm 37.85$ ,  $893.3 \pm 70.8$ , and  $438.5 \pm 130.1 \text{ kg N ha}^{-1} \text{y}^{-1}$  in “Farm land” of CF1, CF2, OF1 and OF2, respectively (Figures 4.46 and 4.47). The N loss via runoff and leaching higher than N input in “Farm land” indicated soil N was being washed off. This is in line with N flow model and N budgeting in Section 4.4 and 4.5 which highlighted potential N surpluses in this study that is vulnerable to leaching and runoff losses. This was supported by Song et al. (2009) who indicated N surplus of more than  $100 \text{ kg N ha}^{-1}$  was susceptible to N loss via volatilization, runoff and leaching. The runoff being recycled were  $492 \pm 62$ ,  $615 \pm 37$ ,  $724 \pm 70$ , and  $413 \pm 238 \text{ kg N ha}^{-1} \text{y}^{-1}$  from “Farm land” to “Catchment pond” in CF1, CF2, OF1 and OF2, respectively. Comparable results were reported by Salo & Turtola (2006) that the N loss via runoff range from 102 to  $504 \text{ kg N ha}^{-1} \text{y}^{-1}$  from grass-cereal-bare fallow-green fallow rotation. No

significant differences ( $P>0.05$ ) were observed in the total runoff N concentrations between organic and conventional farms.

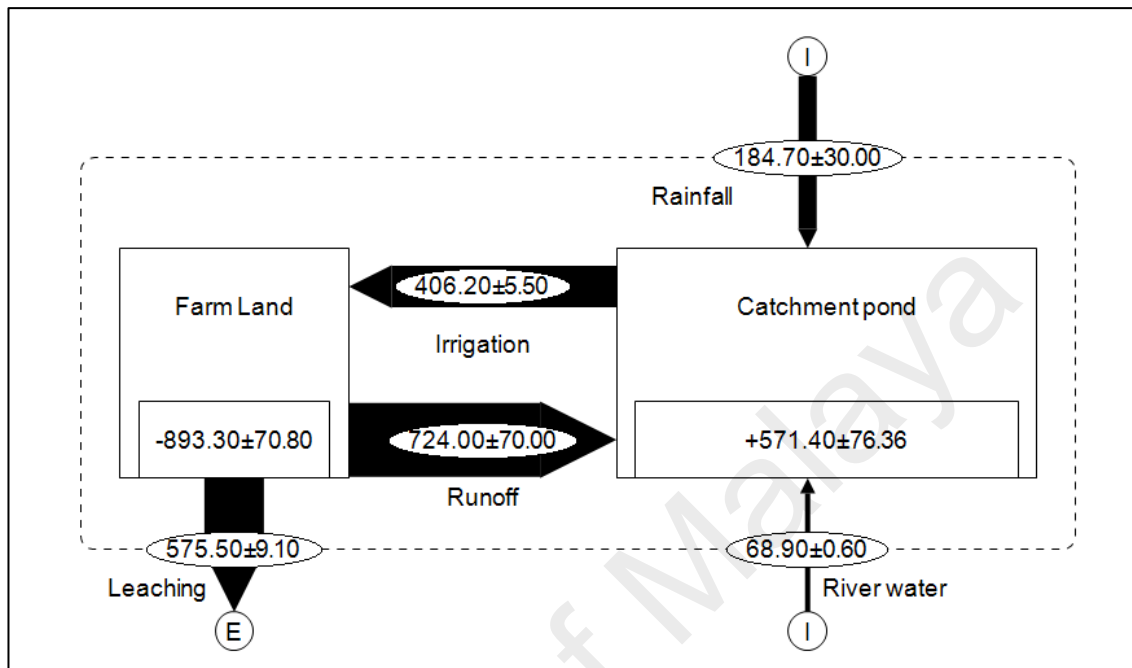


Figure 4.46: N flux through water route in OF1 ( $\text{kg N ha}^{-1} \text{y}^{-1}$ )

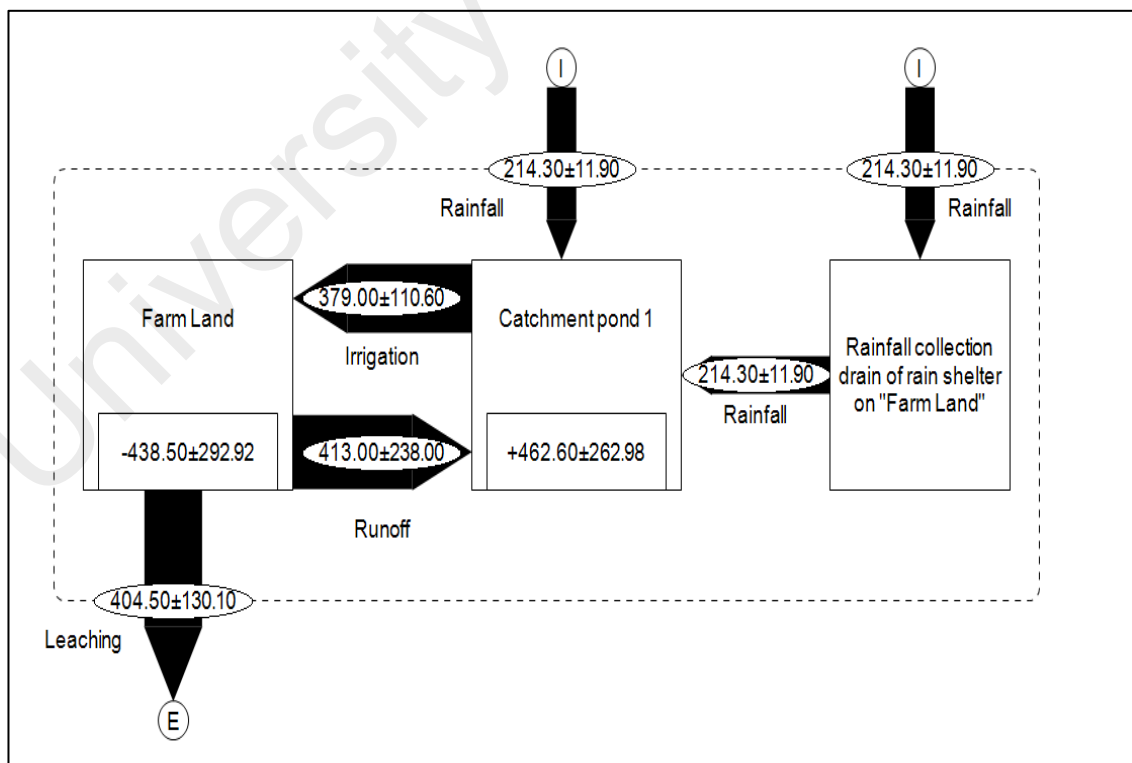


Figure 4.47: N flux through water route in OF2 ( $\text{kg N ha}^{-1} \text{y}^{-1}$ )

The average runoff N concentration was about  $45.1 \pm 42.5$ ,  $51.5 \pm 28.5$ ,  $86.5 \pm 26.8$ , and  $77.4 \pm 38.1$  mg L<sup>-1</sup> in CF1, CF2, OF1 and OF2, respectively. Similar results were reported in runoff water from paddy field where N concentrations range 6.27–62.6 mg L<sup>-1</sup> (Zhao *et al.*, 2012). The author also highlighted the influence of runoff sample collection timing whereby higher N concentration was observed in runoff sample collected after fertilizer application (Zhao *et al.*, 2012). Study showed the amount of N concentrations in runoff often reflects the soil N concentrations (Ramos *et al.*, 2014). However, the soil analysis indicated that the average soil N concentrations were highest in CF2 ( $0.458 \pm 0.011\%$ ) > OF2 ( $0.431 \pm 0.069\%$ ) > OF1 ( $0.3545 \pm 0.012$ ) > CF1 ( $0.209 \pm 0.003\%$ ). The lack of correlation between runoff N and soil N concentrations might be due to the variations in rainfall and irrigation intensities that induced different concentrations of particulate in runoff and leaching (Cade-Menun *et al.*, 2013). In addition, several studies suggested higher correlation between the nutrient concentrations in surface water with fertilizer application and runoff volume than soil properties and nutrient balance (Aweng *et al.*, 2011; Bechmann *et al.*, 2008; Ibrahim *et al.*, 2011; Ismail *et al.*, 2007; Mohd Ekhwan *et al.*, 2012).

The “Catchment pond” in CF1, CF2, OF1, and OF2 was accumulating N by  $499.5 \pm 62.8$ ,  $538.4 \pm 37.7$ ,  $571.4 \pm 76.36$ , and  $240.2 \pm 45.74$  kg N ha<sup>-1</sup> y<sup>-1</sup> during the study. This might lead to drastic increase of vegetation productivity and modification in species composition in “Catchment pond”, as the N loading exceeded the 25 kg N ha<sup>-1</sup> y<sup>-1</sup> critical load (WHO, 2000). However, the N load rates in the farms in this study were below the maximum load of 1000 kg N ha<sup>-1</sup> y<sup>-1</sup> that potentially leads to high N loss via leaching and volatilization (Verhoeven *et al.*, 2006). The irrigation water (CF1:  $9.62 \pm 5.7$ , CF2:  $10.35 \pm 4.66$ , OF1:  $21.2 \pm 1.74$ , OF2:  $24.06 \pm 15.3$  mg L<sup>-1</sup>) recycled from “Catchment pond” contained lower nutrient concentrations than runoff (CF1:  $45.1 \pm 42.5$ ,

CF2:  $51.5 \pm 28.5$ , OF1:  $86.5 \pm 26.8$ , OF2:  $77.4 \pm 38.1$  mg L<sup>-1</sup>) received by “Catchment pond”. This suggested that the “Catchment pond” in this study acted as riparian zone that retained and removed the nutrient in runoff via sedimentation, adsorption and aquatic plants uptake (Chen, 2011; Gao *et al.*, 2015; Qiangl *et al.*, 2005; Verhoeven *et al.*, 2006). The increased N level in “Catchment pond” may cause eutrophication that can increase biomass production and C burial which indirectly contributed to C sequestration (Clow *et al.*, 2015; Dong *et al.*, 2012; Heathcote & Downing, 2012). Due to the labile nature of N compound in water and soil, the N stocks in “Catchment pond” were unlikely to be stored for long period of times. Thus, the N stocks were potentially lost to environment via denitrification, volatilization, runoff or leaching (Bouwman *et al.*, 2013; Ju *et al.*, 2006).

Leaching in farms in this study accounts for N output of  $27.5 \pm 0.2$ ,  $324.1 \pm 3.2$ ,  $575.5 \pm 9.1$ ,  $404.5 \pm 130.1$  kg N ha<sup>-1</sup> y<sup>-1</sup> from CF1, CF2, OF1 and OF2, respectively. Leaching study in Finland indicated N leaching can range between 2-353 kg N ha<sup>-1</sup> y<sup>-1</sup> and the N leaching was highly variable due to differences in farm management (Salo & Turtola, 2006). In addition, a research demonstrated an increase of total N input will lead to higher N leaching, eg. 1,254 and 3,154 kg N ha<sup>-1</sup> y<sup>-1</sup> of N input resulted in 217 and 569 kg N ha<sup>-1</sup> y<sup>-1</sup> of N leaching, respectively (Zhu *et al.*, 2005). Research suggested not all N loss from agricultural land will enter the water bodies as it may be retained by the soil particle along the transport route (Bashir *et al.*, 2013; Reidsma *et al.*, 2012). Therefore, transport factors like soil texture, soil permeability, cation exchange capacity (CEC), saturated hydraulic conductivity, slope of cultivation area, distance of agriculture land to water body, riparian buffer zone, irrigation erosion, rainfall, surface runoff, leaching, soil erosion, drainage and biological activity will determine the extent of nutrient loss

entering the water bodies (Nie *et al.*, 2012; Pärn *et al.*, 2012). Thus, the results only reflect potential N loss from farm systems that might enter the water systems.

The average water outflow volume in conventional systems (CF1: 14,735 t ha<sup>-1</sup> y<sup>-1</sup> and CF2: 15,925 t ha<sup>-1</sup> y<sup>-1</sup>) were larger than organic systems (OF1: 10,721 t ha<sup>-1</sup> y<sup>-1</sup> and CF2: 7,253 t ha<sup>-1</sup> y<sup>-1</sup>). However, high N loss was observed in organic systems due to higher N concentrations in runoff and leachate. This suggested a great influence of fertilizer regime, soil biological activity and N mineralization process over the N water flux (Ramos *et al.*, 2014). Therefore, factors such as complex biological, physical, and chemical processes, farm management practices, climatic conditions and soil properties are governing the N loss (Congreves & Van Eerd, 2015).

The water flow models highlighted the importance of water management and strategies in minimizing N loss and increased nutrient and water use efficiencies. Study suggested the increased of soil available N will induce higher N loss potential through runoff. However, it depends on water transport from irrigation and rainfall (Korsaeth & Eltun, 2000). Several authors supported this and highlighted the reduction of irrigation volume can potentially reduce the nutrient and C loss from farm soils (Fang *et al.*, 2010a; Fang *et al.*, 2010b).

#### **4.7.4 Nitrate in runoff and leaching**

About 30-45% and 25-60% of total N in runoff and leachate are of nitrate. The nitrate concentrations in runoff range from 16.5 ± 10.7 mg L<sup>-1</sup> in CF2 to 29.6 ± 23.6 mg L<sup>-1</sup> in OF2 while in leachate it range from 1.75 ± 0.55 mg L<sup>-1</sup> in CF1 to 95.7 ± 17.1 mg L<sup>-1</sup> in OF1 (Figures 4.48 and 4.49). Similar results were observed in Zhao *et al.* (2010) and

Song et al. (2009) which reported leachate nitrate concentrations range from 43.1 to 74.2 mg L<sup>-1</sup> and 17 to 457 mg L<sup>-1</sup>, respectively.

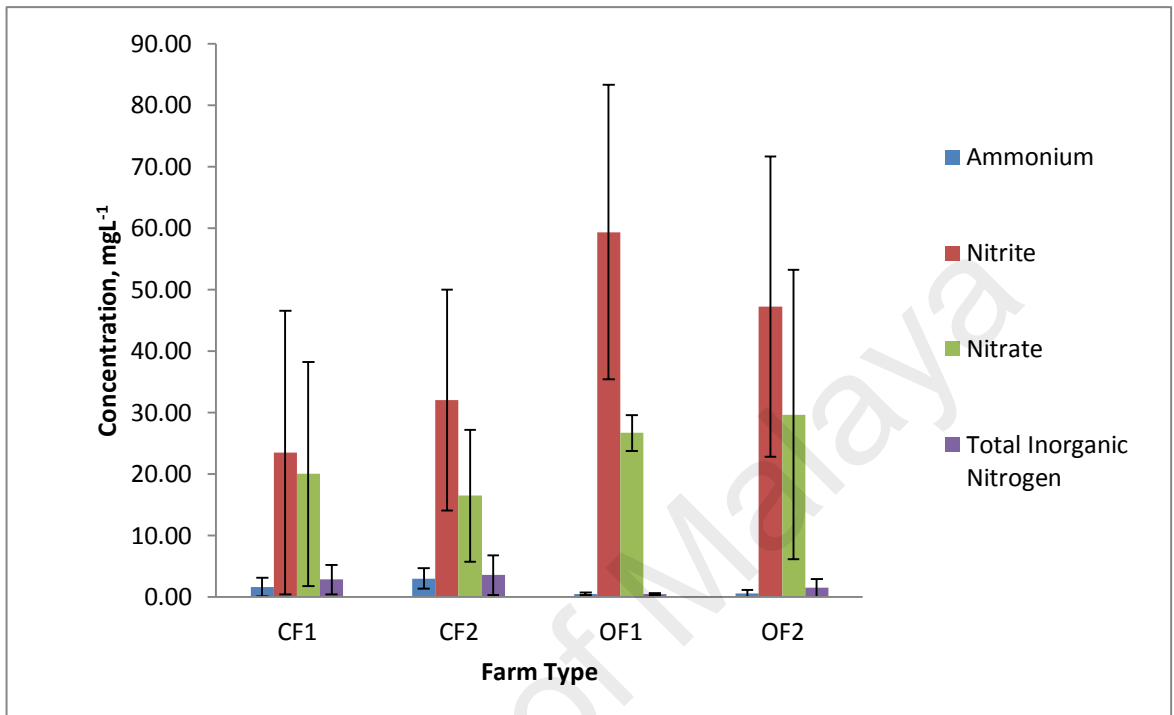


Figure 4.48: Ammonium, nitrite, nitrate and total inorganic N concentrations in runoff

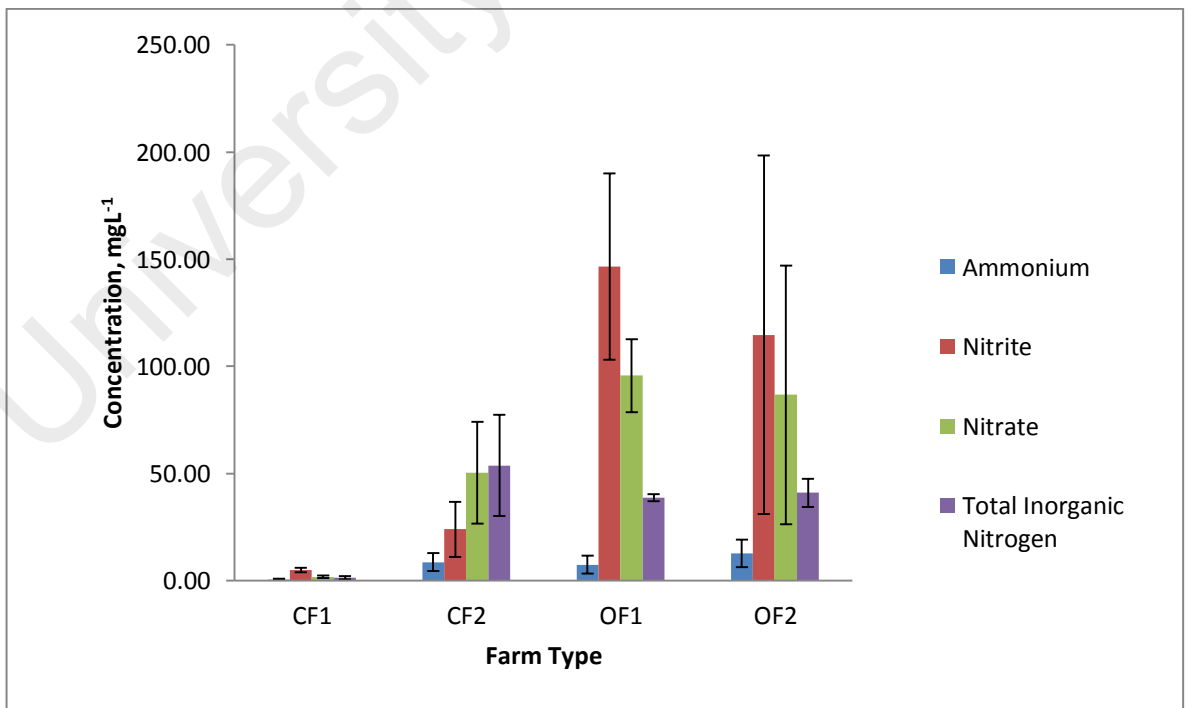


Figure 4.49: Ammonium, nitrite, nitrate and total inorganic N concentrations in leachate



Based on the EPA's 10 mg L<sup>-1</sup> nitrate concentrations limit for drinking water (WHO, 2011b), almost all the runoff and leachate samples in this study exceeded the limit except for leachate samples in CF1. However, the nitrate concentrations limit for drinking water in European countries (EC) is higher than EPA limit which is 50 mg L<sup>-1</sup> (DEFRA, 2015b). Based on EC limit only leachate samples in CF2, OF1 and OF2 exceed the limit. The high nitrate concentrations in runoff and leachate will be detrimental to water quality if it enters the ground and surface water (Esmaeili *et al.*, 2014; Ross, 2010).

There is a significantly higher nitrate concentrations (P=0.011) in leachate of organic farms (357 mg L<sup>-1</sup>) in comparison to conventional farms (37.5 mg L<sup>-1</sup>). However, no difference was observed for runoff nitrate (P>0.05). Similar results were reported by Ramos *et al.* (2014) which the average soluble nitrate concentrations in organic systems (approximately 38.62 mg L<sup>-1</sup>) were eight times higher than conventional systems which resulted in five times higher nitrate loss in the organic systems. Variation in fertilizer regime might be the main cause of different nitrate concentrations in the farming systems (Dahan *et al.*, 2014). Several researches showed higher available N in organic farms as compared to conventional farms (Addiscott & Benjamin, 2004; Burger & Jackson, 2003; De Vries *et al.*, 2006; Stark *et al.*, 2008). This is because the high organic matter applications in organic system encouraged microbial decomposition rate that immobilized soil N and recycled nitrate (Gaskell & Smith, 2007). Compost has slow and stable N release rate and study showed after 32 weeks of compost application about 11-29% of compost N were still being released (Duong *et al.*, 2013; Hadas & Portnoy, 1994). Other than that, the organic farms were cultivated under plastic film greenhouse that modified the soil water balance resulting from lack of rainfall leaching and strong evaporation of soil water which leads to accumulation of soil nitrate (Scotti

*et al.*, 2015a). This was supported by Ju *et al.* (2007) who reported 67-76% of total anion in salinity soil was nitrate and greenhouses soil salinization is characterized by nitrate accumulation in soil.

#### **4.8 Greenhouse gas (GHG) emissions**

Gaseous emission from agriculture soils was assessed over the study period. The objective was to quantify C and N emissions from gaseous emit from arable soil. Three chambers were deployed during each measurement and were installed before measurement to avoid any obstruction to field operation (e.g. fertilizer, pesticide and herbicide application, tillage, and harvesting). Overall, significant differences were found in each farm but no differences were observed between organic and conventional farm systems. Results suggested gaseous emissions were highly variable and dependent on individual farm management. Gaseous emissions exceeding 100% variability were often observed in field due to complex link of factors that influenced flux rate (Sainju *et al.*, 2012). Gaseous flux rate variation can be due to methodology, chamber size, chamber type (static or portable), and chamber placement (Hénault *et al.*, 2012). In addition, the quantification of gaseous emissions from agriculture soil is highly dependent on various factors such as climate, soil management, soil properties (texture, moisture content, and organic matter), vegetation, landscape and time (Bashir *et al.*, 2013).

##### **4.8.1 Ammonia (NH<sub>3</sub>)**

Ammonia concentration in the static chamber increased gradually during the measurement (Figure 4.50). The initial NH<sub>3</sub> concentrations in CF1 and OF1 were 20.4 ± 8.3 and 19.3 ± 11.1 ppm which was higher in comparison to CF2 and OF2 that range from 0 - 0.41 ppm. The NH<sub>3</sub> volatilization varies among each study farms ( $P < 0.01$ ).

However, there is no statistical difference between organic system and conventional systems ( $P > 0.05$ ).

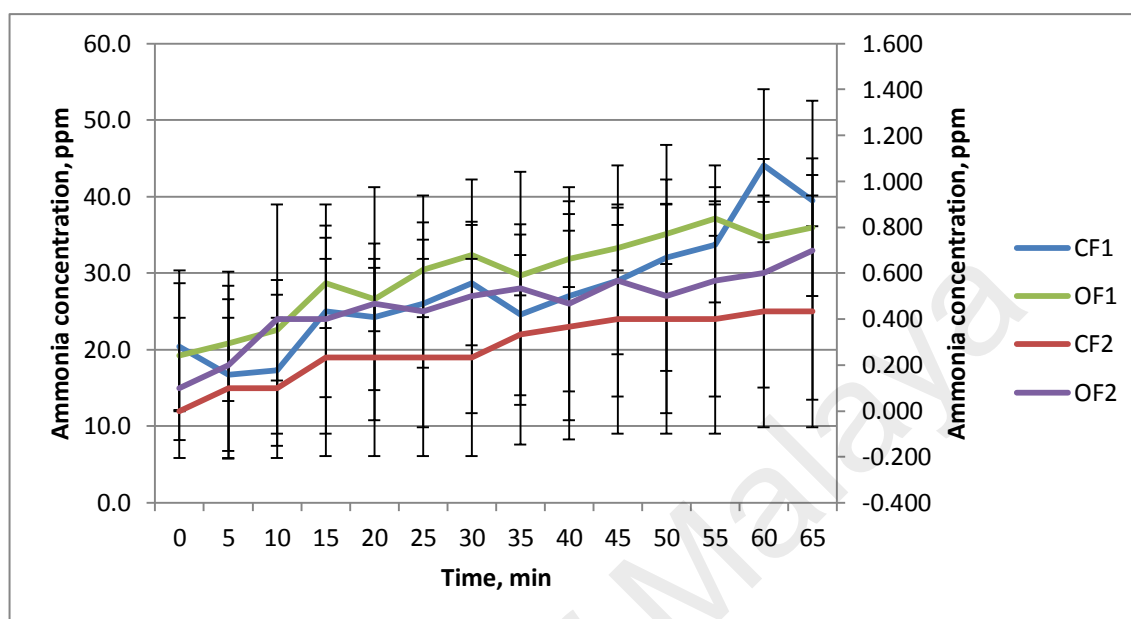


Figure 4.50: NH<sub>3</sub> concentration in static chamber over time

The average NH<sub>3</sub> volatilization rate in CF1, CF2, OF1 and OF2 was  $38 \pm 18.9$ ,  $1.03 \pm 0.99$ ,  $35.89 \pm 5.52$  and  $1.06 \pm 0.98 \mu\text{L NH}_3 \text{ L}^{-1} \text{ min}^{-1}$  which resulted N outputs of 113, 3, 107 and 16 kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively (Figure 4.51). The results were supported by Saggari et al. (2004), who emphasized that the direct application of manure (101–1,100 kg N ha<sup>-1</sup>) to pasture soil may lead to an N loss of 17–316 kg N ha<sup>-1</sup> y<sup>-1</sup> via NH<sub>3</sub> volatilization. Results similar to CF1 and CF2 were reported in soil with different urea application rate via subsurface banding method which range between 131.4 and 175.1 kg N ha<sup>-1</sup> y<sup>-1</sup> ( $1.5\text{--}2 \text{ mg N m}^{-1} \text{ h}^{-1}$ ) (Rochette *et al.*, 2013). Lower NH<sub>3</sub> volatilization of 15 kg N ha<sup>-1</sup> was reported by Zhu et al. (2005) with sponge absorption method. The NH<sub>3</sub> volatilization in CF1 and CF2 were mainly from chicken manure application. Study showed soil fertilized with chicken manure can contain high amount of NH<sub>4</sub><sup>+</sup> (as high as 89% of total soil N) that can be lost to the environment via volatilization or leaching (Warman & Cooper, 2000).

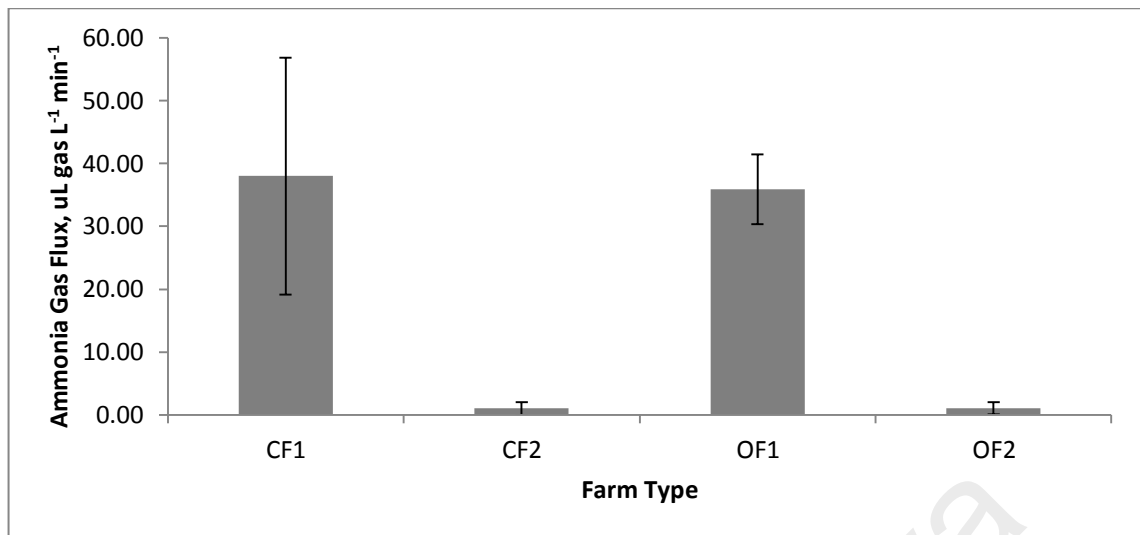


Figure 4.51: NH<sub>3</sub> gas flux rate in the farming systems

Soil ammonium, nitrification and denitrification processes in agricultural soils were the main sources for NH<sub>3</sub> volatilization (Yli-Viikari *et al.*, 2007). Ammonia especially in the topsoil layer can be easily volatilized and released to the atmosphere (Jarvis *et al.*, 2011; Pärn *et al.*, 2012; Tilman, 1999). Thus, the N input via application of soil amendments and fertilizer influenced the NH<sub>3</sub> volatilization (He *et al.*, 2003). The NH<sub>3</sub> volatilization rate in CF1 was generally higher than CF2. Study showed an increased of soil pH markedly increased NH<sub>3</sub> volatilization from soil (Ernst & Massey, 1960; Fenn & Kissel, 1973; Rochette *et al.*, 2013). Thus, the slightly acidic to neutral soil in CF2 might contribute to lower NH<sub>3</sub> volatilization to CF1 which has neutral to moderately alkaline soil (See section 4.1.1 for soil pH classification). Other than that higher volatilization rate in CF1 might also due to the co-application of chicken manure, compost and synthetic fertilizer in CF1 while CF2 was only applied with chicken manure and synthetic fertilizer.

Research indicated co-application of compost and synthetic fertilizer significantly increased the rate of NH<sub>3</sub> volatilization (Matsushima *et al.*, 2009). However, a research reported composts with high C:N ratio (> 20:1) and high C content can stimulate

microbial immobilization and reduce soil available N that are susceptible to loss (Gaskell & Smith, 2007). The compost used in all the farms in this study have C:N ratio that range from 8:1 to 12:1. The low soil C:N in farms in this study indicated soil N are susceptible to N mineralization (Haney *et al.*, 2012; Huang *et al.*, 2004). In addition, study suggested urea hydrolysis potential in compost is one of the important factor that induced NH<sub>3</sub> volatilization (Matsushima *et al.*, 2009). Thus, variation of urea hydrolysis potential in composts might be the reason for higher NH<sub>3</sub> volatilization in OF1 as compared to OF2. Other than that, several authors reported heavy fertilized and irrigated soil have higher potential to NH<sub>3</sub> volatilization (Gaskell & Smith, 2007; Zhang *et al.*, 2014). Therefore, higher total N input in OF1 ( $2,718 \pm 7.35 \text{ kg N ha}^{-1} \text{ y}^{-1}$ ) coupled with large amount of water input ( $19,283 \pm 3,148 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ ) in comparison to OF1 (total N input:  $1,349 \pm 111.9 \text{ kg N ha}^{-1} \text{ y}^{-1}$ ; water input:  $15,815 \pm 7,260 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ ) might also lead to high NH<sub>3</sub> emission in OF1.

Figure 4.52 shows higher NH<sub>3</sub> emission in first two weeks of planting stage (fertilizer was applied one the day before planting) and decreased during third and fourth week of planting in CF1 and OF1. Similar NH<sub>3</sub> volatilization trends were reported by Rochette *et al.* (2013) in which increased of NH<sub>3</sub> volatilization rates from day one ( $6.1 \text{ g N m}^{-2}$ ) to day nine ( $15.3 \text{ g N m}^{-2}$ ) after soil fertilization. Similarly, Peng *et al.* (2015) also reported 70% of total NH<sub>3</sub> emission occurred in less than 10 days after fertilization. However, no statistical differences were observed in NH<sub>3</sub> volatilization ( $P=0.744$ ) of each planting stages. The irregular NH<sub>3</sub> volatilization in farms in this study might be due to the variation of soil pH, soil texture, soil CEC, soil moisture, wind velocity, temperature, climate and anthropogenic disturbances (Ernst & Massey, 1960; Li, 2000; Rochette *et al.*, 2013; Sagar *et al.*, 2004; Salo & Turtola, 2006).

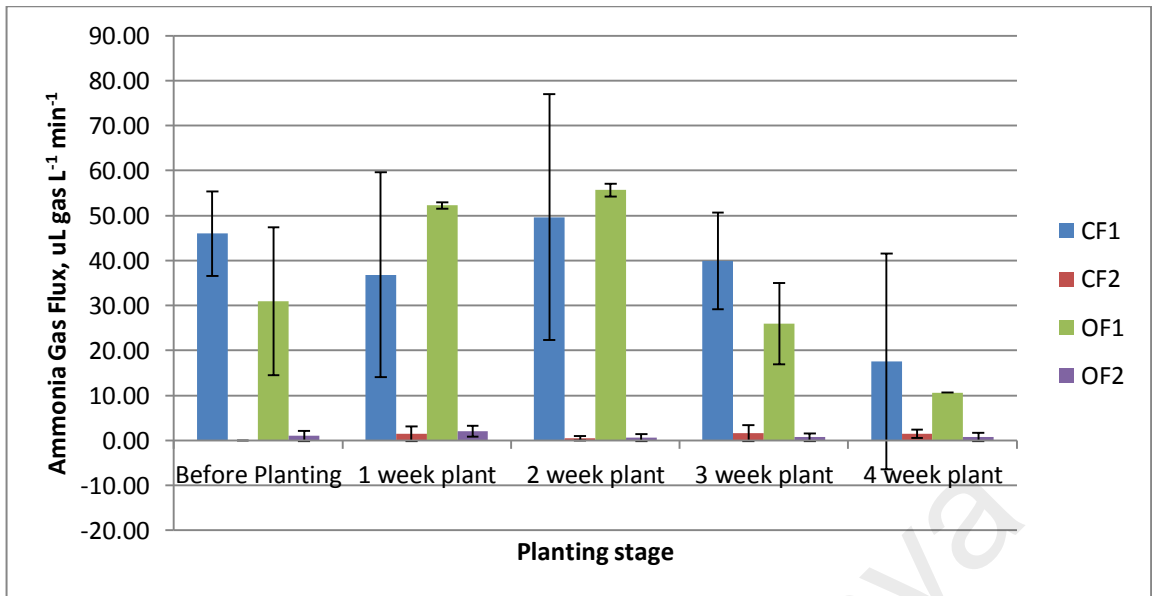


Figure 4.52: NH<sub>3</sub> gas flux at different planting stage

#### 4.8.2 Carbon dioxide (CO<sub>2</sub>)

The average initial CO<sub>2</sub> concentrations during field study were about  $333 \pm 617$ ,  $153 \pm 148$ ,  $1,267 \pm 1,033$  and  $187 \pm 242$  ppm and the concentration increased steadily over the time (Figure 4.53). There was a significant difference observed in CO<sub>2</sub> emissions between each farm ( $P < 0.01$ ). However, no differences were detected in CO<sub>2</sub> ( $P = 0.302$ ) emissions between each planting stage (Figure 4.54).

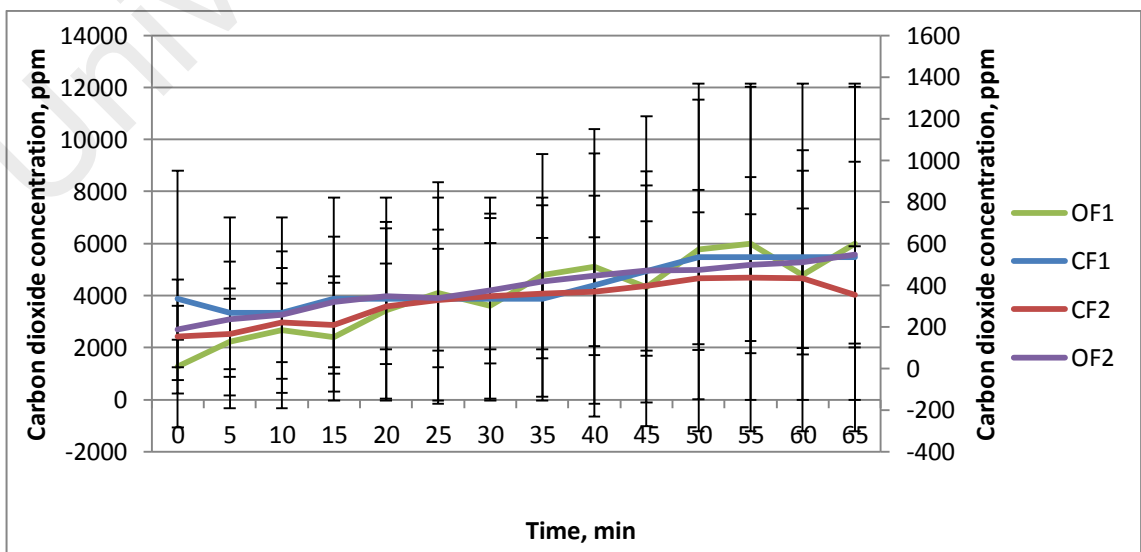


Figure 4.53: CO<sub>2</sub> concentration in static chamber over time

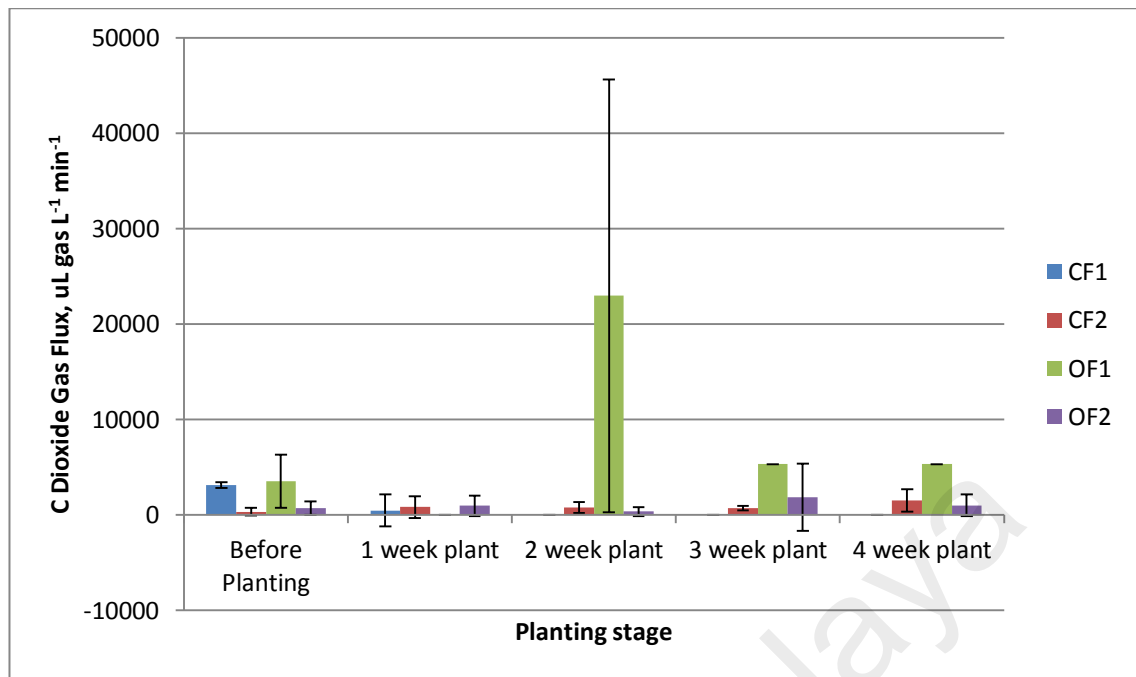


Figure 4.54: CO<sub>2</sub> gas flux at different planting stage

The CO<sub>2</sub> gaseous flux rate was around  $717 \pm 390$ ,  $841 \pm 709$ ,  $7,432 \pm 5,095$  and  $990 \pm 1,372$   $\mu\text{L CO}_2 \text{ L}^{-1} \text{ min}^{-1}$  which contributed to an average C output of 1,830, 2,150, 18,916, 2,534  $\text{kg C ha}^{-1} \text{ y}^{-1}$  from CF1, CF2, OF1 and OF2, respectively (Figure 4.55). Results comparable to CF1, CF2 and OF2 were reported by a six months closed chamber study on two vegetable cultivating sites in which the CO<sub>2</sub> flux range 2,703-3,507  $\text{kg C ha}^{-1}$  and 5,541-7,324  $\text{kg C ha}^{-1}$  (Mu *et al.*, 2013). A study on malt-barley plot with tillage practices indicated an average CO<sub>2</sub> flux of 13,432  $\text{kg C ha}^{-1} \text{ y}^{-1}$  (36.8  $\text{kg C ha}^{-1} \text{ d}^{-1}$ ) measured with portable chamber was about (Sainju *et al.*, 2012). However, a meta-analysis study of tall grass prairie showed CO<sub>2</sub> flux range from 10,000 to 21,000  $\text{kg C ha}^{-1} \text{ y}^{-1}$  between year 1993 and 1998 and the author highlighted the seasonal, temperature and soil moisture differences can influence the average annual C emissions (Mielnick & Dugas, 2000). Thus, the different results of CO<sub>2</sub> emissions rate from agriculture soil might be due to the variations in soil properties (temperature, moisture content, organic matter and texture), climate, vegetation, landscape position and microbial activity (Poll *et al.*, 2013).

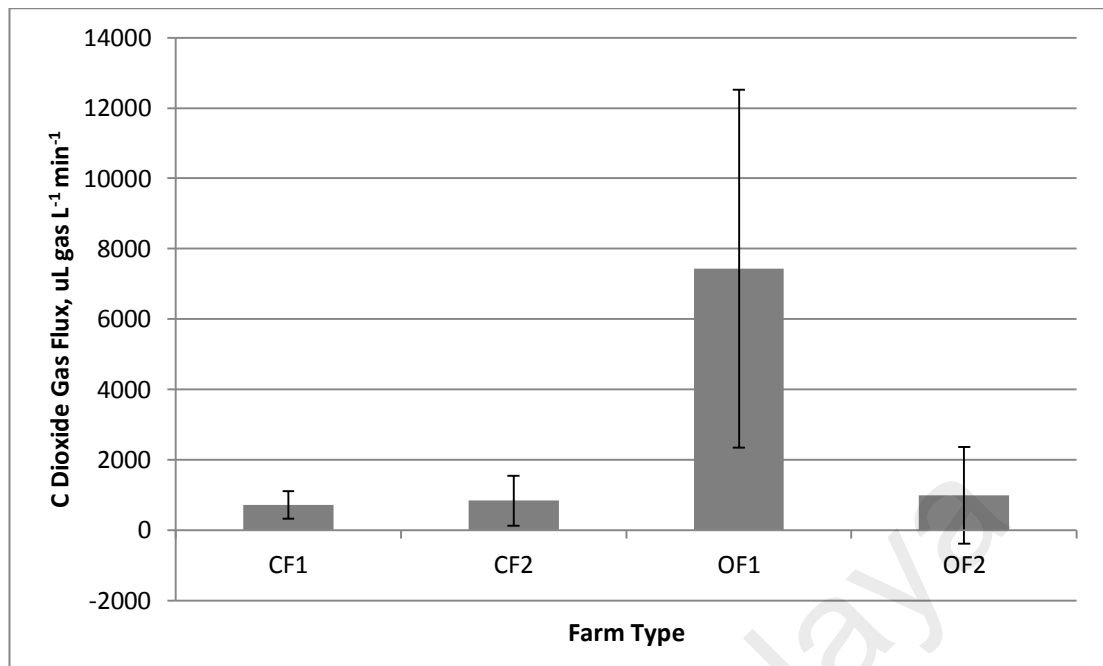


Figure 4.55: CO<sub>2</sub> gas flux rate in the farming systems

The results showed the CO<sub>2</sub> emissions from organic systems were higher than in conventional systems ( $P = 0.05$ ). The compost input in the organic farm systems might induce higher CO<sub>2</sub> emissions as compared to conventional systems. Soil respiration is the primary pathway of CO<sub>2</sub> loss from agriculture soil due to root and microbial activity (Schlesinger & Andrews, 2000). Therefore, organic material input such as compost and the decomposition rates determined the net CO<sub>2</sub> emissions from soils (Scotti *et al.*, 2015a; Setia *et al.*, 2011; Yli-Viikari *et al.*, 2007). Study by Jaiarree *et al.* (2014) demonstrated the correlation between C emissions from soil and compost application rate which the compost application rate of 30,000 and 50,000 kg ha<sup>-1</sup> y<sup>-1</sup> resulted C flux of 9,860 and 10,140 kg C ha<sup>-1</sup> y<sup>-1</sup>, respectively. Thus, the high organic matter (compost) input of 206,850 kg ha<sup>-1</sup> y<sup>-1</sup> (4 to 8 times higher than CF1, CF2 and OF2) in OF1 might induce higher CO<sub>2</sub> emissions.

In addition, the Bokashi compost (18,750 kg ha<sup>-1</sup> y<sup>-1</sup>) used in OF1 contained effective microbes (EM). Studies showed compost with EM can stimulate the degradation of



organic materials and accelerate C mineralization process and the soil with EM respired 38% more C than the soil without EM (Daly & Stewart, 1999; Hu & Qi, 2013). This is in line with the aerobic microbial plate count results (Section 4.9.1) in this study which suggested greater soil microbial population in OF1. Therefore, the high compost application coupled with increased microbial activity might be the reason for the high CO<sub>2</sub> emissions in OF1.

Soil C availability is the main limiting factor for microbial growth and activity thus the increased C supply encouraged soil microbial activity which leads to CO<sub>2</sub> emissions (Setia *et al.*, 2011). In addition, the C:N ratio of soils and composts used in farms in this study were below 20:1-30:1 threshold suggested higher susceptibility to CO<sub>2</sub> emissions (Haney *et al.*, 2012; Huang *et al.*, 2004). The increased of SOM and microbial decomposition rate can induced greater CO<sub>2</sub> emissions (Killebrew & Wolff, 2010; Sainju *et al.*, 2012; Schlesinger & Andrews, 2000; Stockmann *et al.*, 2013; Tilman, 1999).

#### **4.8.3 Carbon monoxide (CO)**

The average initial CO concentrations recorded in CF1, CF2, OF1 and OF2 was  $16.3 \pm 2.37$ ,  $0.333 \pm 0.959$ ,  $16.8 \pm 3.91$  and  $0.63 \pm 1.75$  ppm, respectively and the concentration fluctuated during the static chamber measurement (Figure 4.56). There were significant differences in CO emissions detected between each farm ( $P < 0.01$ ). However, no differences were observed between the organic and conventional systems ( $P > 0.05$ ) and between planting stages ( $P=0.612$ ) (Figure 4.57).

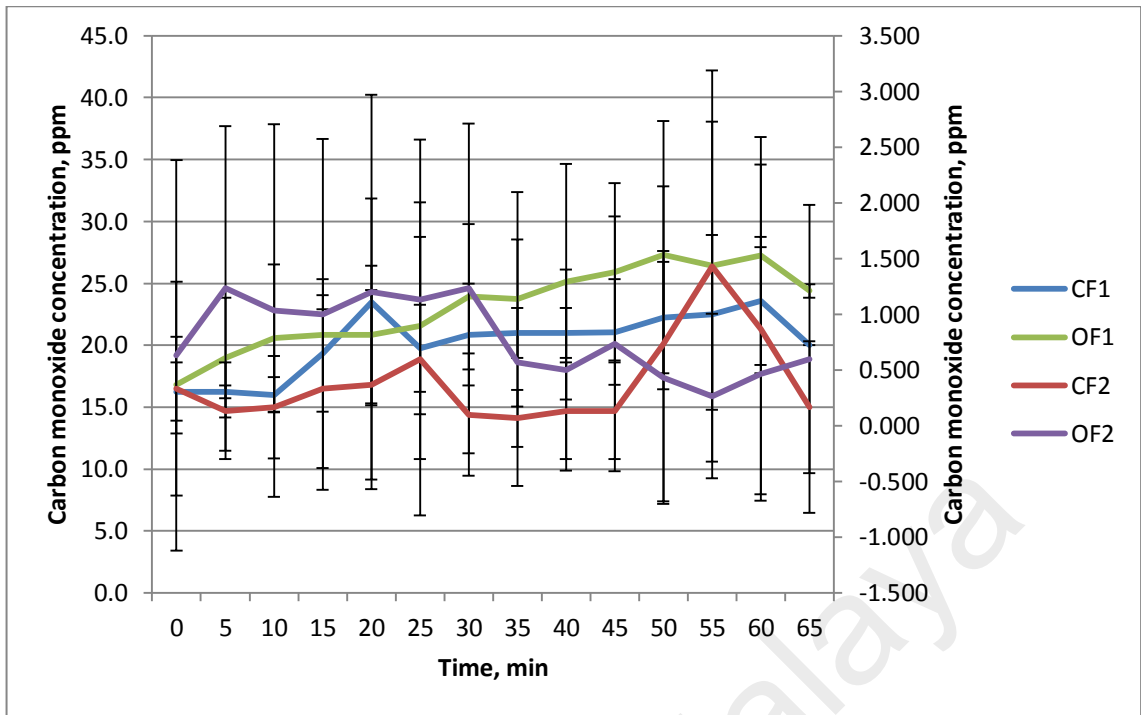


Figure 4.56: CO concentration in static chamber over time

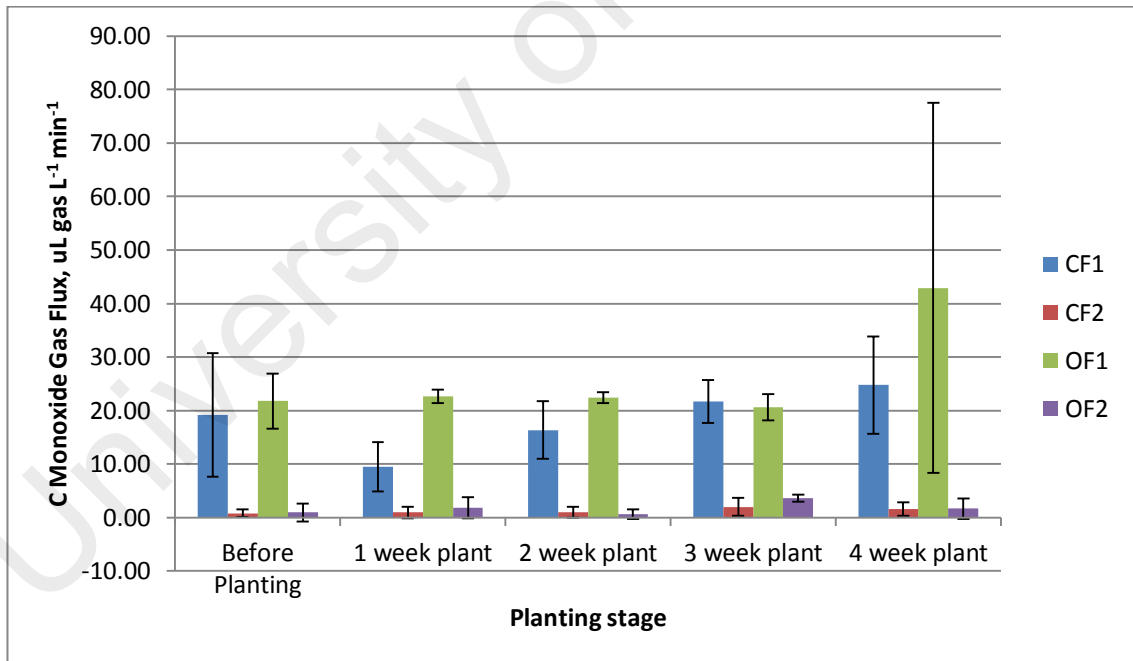


Figure 4.57: CO gas flux at different planting stage

The CO emissions rate of  $18.29 \pm 6.94$ ,  $1.23 \pm 1.15$ ,  $26.06 \pm 8.92$  and  $1.72 \pm 1.42$   $\mu\text{L CO L}^{-1} \text{ min}^{-1}$  had resulted in C mass flux of 47, 3.2, 66, and 4.4  $\text{kg C ha}^{-1} \text{ y}^{-1}$  in CF1, CF2, OF1 and OF2, respectively (Figure 4.58).

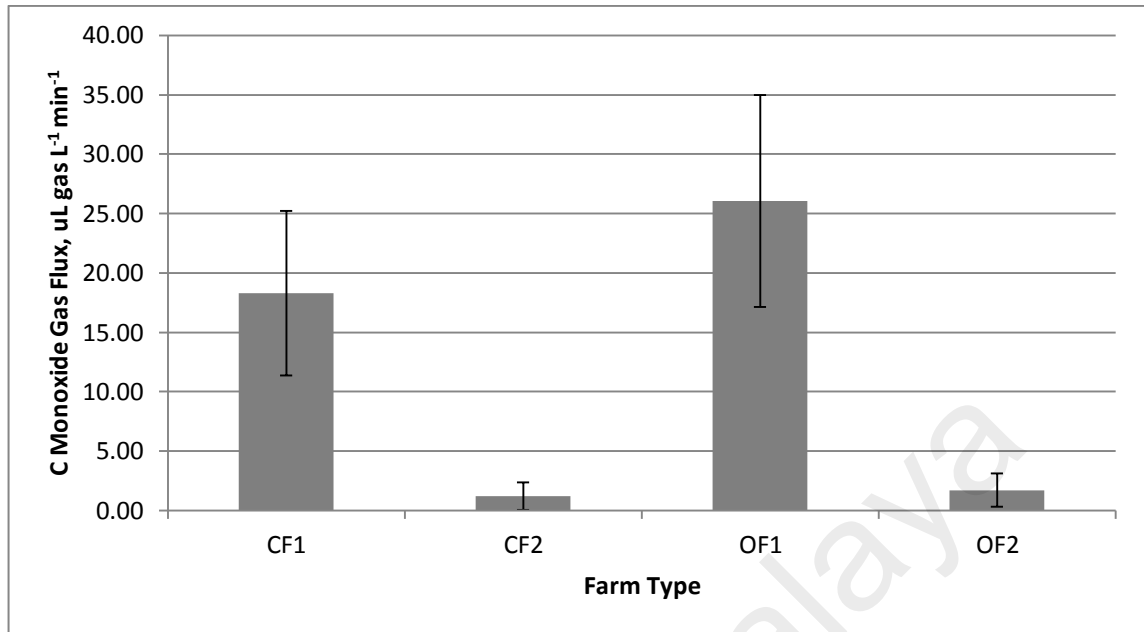


Figure 4.58: CO gas flux rate in the farming systems

The contribution of C mass outputs through CO emissions was relatively small as compared to CO<sub>2</sub> emissions. This was supported by Olivier et al. (1999) which highlighted that the contributions of CO emissions from vegetation is small as compared to anthropogenic activities and little data is available for CO emissions from vegetable cultivating soil.

The factors controlling CO emissions from vegetation and soil were not well known (Guenther *et al.*, 2000). There were several theories of the CO flux from soil. Some authors suggested non-biological decomposition of humic materials, chemical oxidation of soil organic C, or even anaerobic microbial activity were the processes may lead to CO emissions from soil (Conrad & Seiler, 1980; Moxley & Smith, 1998). Researches demonstrated that the CO production rates were significantly correlated to soil respiration, total C, N and NH<sub>4</sub><sup>+</sup> content in soil in which higher CO flux was observed from soil with high organic C (eg. CO flux in forest soil is higher than arable land)

(Gödde *et al.*, 2000; Yonemura *et al.*, 2000). Thus, higher CO emissions from OF1 as compared to other farms in this study might be due to high soil C content and total C input. Other than that, the emissions of CO during the cropping cycle might be from plant photochemical reaction. Study suggested CO emissions from living plant was mainly due to photochemical transformation inside the leaf (Guenther *et al.*, 2000).

#### 4.8.4 Methane (CH<sub>4</sub>)

CH<sub>4</sub> emissions were only detected in CF2 and OF2 and there was significant differences in CH<sub>4</sub> emissions between each farm ( $P < 0.01$ ). However, no significance differences were detected between the organic and conventional systems ( $P > 0.05$ ) and also between the different planting stages for CH<sub>4</sub> emission ( $P = 0.141$ ) (Figure 4.59).

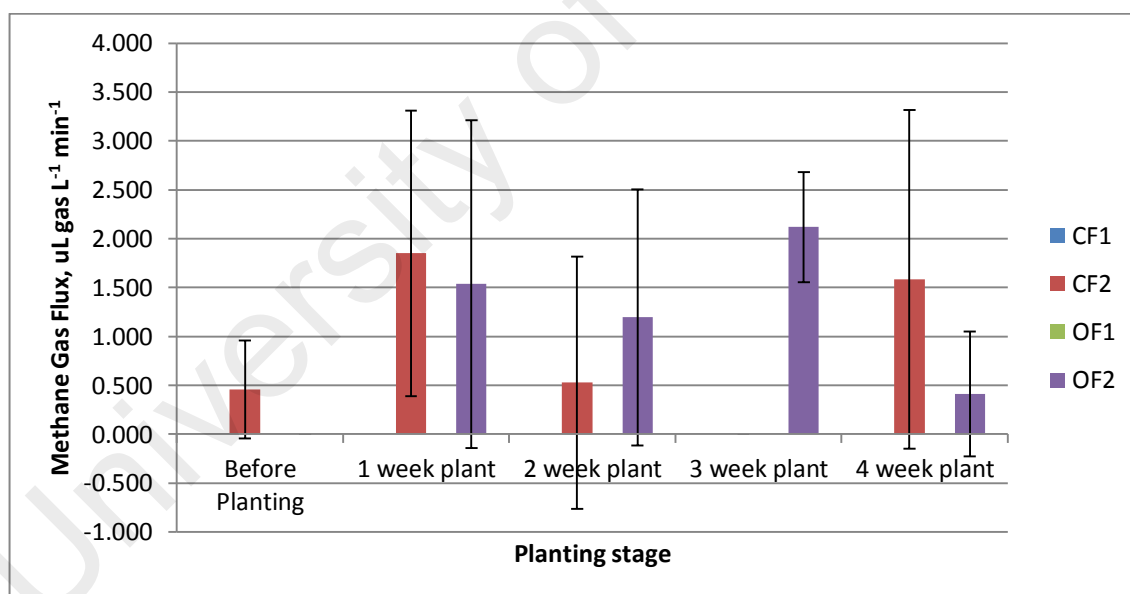


Figure 4.59: CH<sub>4</sub> flux at different planting stage

During the measurement of gaseous, the average CH<sub>4</sub> concentrations were increased in the static chamber with the initial concentrations of 0 ppm in both CF2 and OF2 (Figure 4.60). Methane gaseous flux rate in CF2 and OF2 was  $0.8838 \pm 0.9972$  and  $1.052 \pm 0.838 \mu\text{L CH}_4 \text{ L}^{-1} \text{ min}^{-1}$ , respectively (Figure 4.61). The methane emissions in CF2 and

OF2 have contributed to C output 2.3 and 2.7 kg C ha<sup>-1</sup> y<sup>-1</sup>, respectively. Similar results were reported by Pohl et al. (2014) that the annual CH<sub>4</sub> flux rates were generally very low (< 3 kg C ha<sup>-1</sup> y<sup>-1</sup>). This was supported by Glatzel & Stahr (2001) who recorded low CH<sub>4</sub> flux of 1.84 and 1.23 kg C ha<sup>-1</sup> y<sup>-1</sup> in unfertilized and fertilised grassland. Low CH<sub>4</sub> emissions were also observed in long term no till cropping system that range from -0.05 ± 0.61 to +0.994 ± 0.105 kg C ha<sup>-1</sup> y<sup>-1</sup> (Bayer *et al.*, 2012). However, the intensive vegetable fields in China demonstrated higher CH<sub>4</sub> emissions that range from 9.0 ± 3.5 to 18.8 ± 1.2 kg C ha<sup>-1</sup> y<sup>-1</sup> (Jia *et al.*, 2012b). A two years study of four different cropping system also demonstrated higher C output via CH<sub>4</sub> emissions which were about 13.90, 18.80, 12.11 and 16.08 kg C ha<sup>-1</sup> (average is 6.95, 9.4, 6.055, 8.04 kg C ha<sup>-1</sup> y<sup>-1</sup>) (Jia *et al.*, 2012a). The CH<sub>4</sub> flux in agriculture soil is often negligible as compared to annual CO<sub>2</sub> flux and supported by Akiyama et al. (2013) and Wu et al. (2015). Different results reported might be due to variations in methodology, time scale, spatial, and environment parameters (Glatzel & Stahr, 2001).

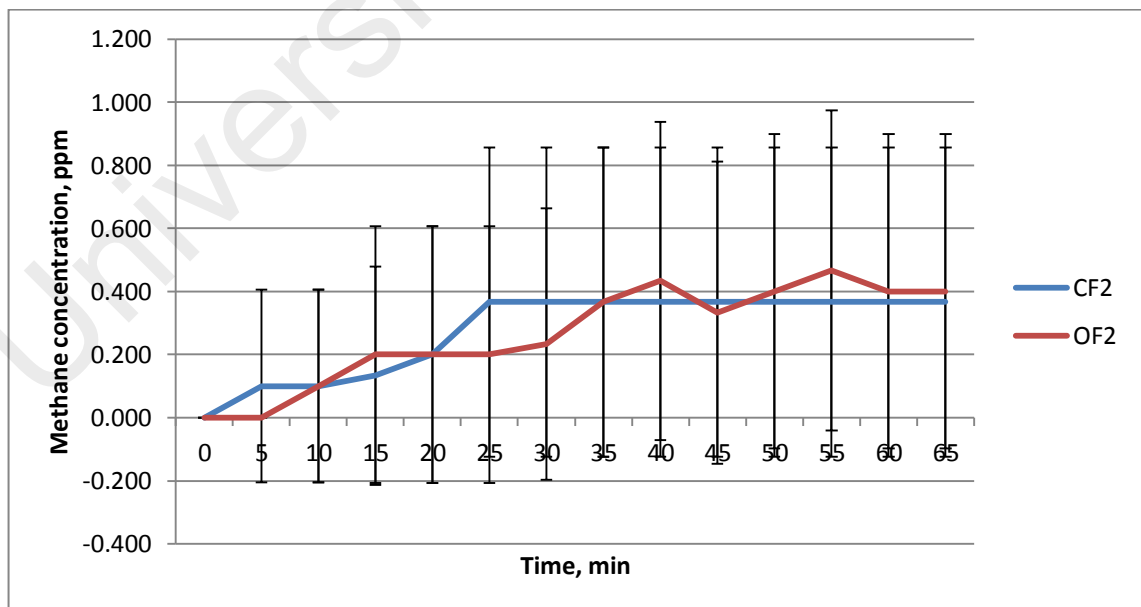


Figure 4.60: CH<sub>4</sub> concentration in static chamber over time

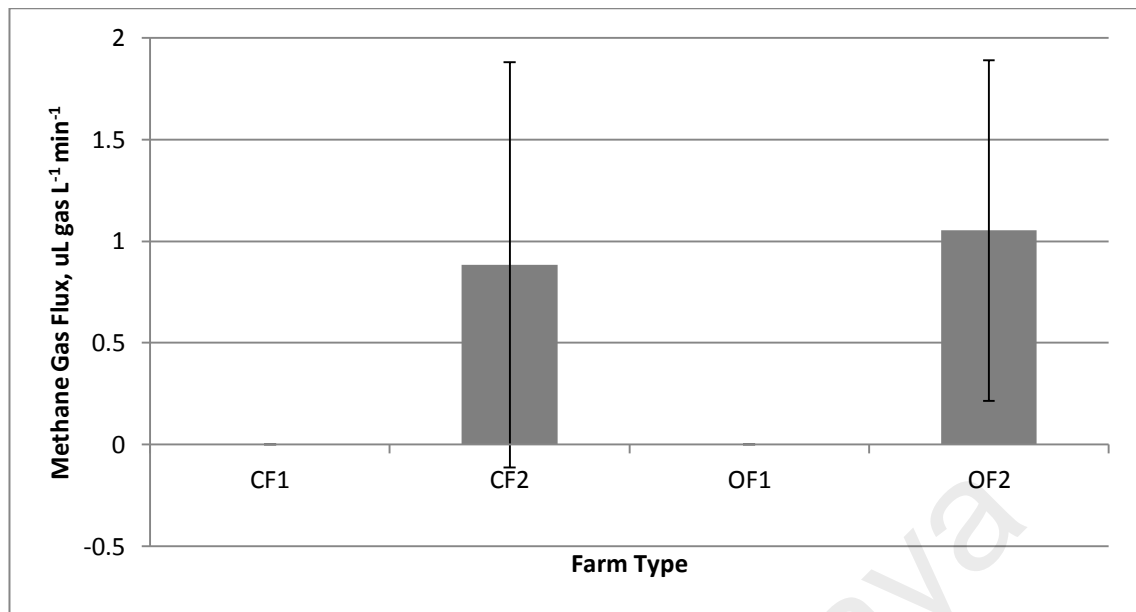


Figure 4.61: CH<sub>4</sub> flux rate in the farming systems

The source of CH<sub>4</sub> in CF2 and OF2 might be from the decomposition of organic matters (chicken manure and compost) especially under anaerobic condition (DEFRA, 2014a; Ginting *et al.*, 2003; Scotti *et al.*, 2015a). Study indicated an increase of organic materials input to soil would considerably increase the CH<sub>4</sub> emissions (Jia *et al.*, 2012b). Study suggested that CH<sub>4</sub> emissions might not necessarily require an anaerobic environment in the entire soil but in small soil aggregates (Glatzel & Stahr, 2001; Megonigal & Guenther, 2008). This was supported by Bayer *et al.* (2012), in which CH<sub>4</sub> emissions was observed in long term no till grass-legume cultivating soil. The author suggested the increase of soil NH<sub>4</sub><sup>+</sup> and DOC content may suppress CH<sub>4</sub> oxidation and stimulates methanogenesis. The activity of CH<sub>4</sub> oxidizers and methanogens in soil leads to high variability in CH<sub>4</sub> emissions rates which can range from -15.61 to 11.60 kg C ha<sup>-1</sup> y<sup>-1</sup> (Bayer *et al.*, 2012; Elder & Lal, 2008).

Methane emission was not observed in CF1 and OF1 because the activity of CH<sub>4</sub> oxidizers might exceed that of methanogens. Study indicated that agriculture land contained highest population of methanotrophs (CH<sub>4</sub> oxidizer) as compared to other soil

such as grassland, forest, landfill, compost soil and saline soil (Akiyama *et al.*, 2014; Tiwari *et al.*, 2015). This was supported by Megonigal & Guenther (2008) who reported the methane produced in soils can be consumed by CH<sub>4</sub> oxidizers without being emitted to the atmosphere.

#### 4.8.5 Nitrous oxide emissions (N<sub>2</sub>O)

The emissions of N<sub>2</sub>O from managed soil, leaching, and runoff are common in agricultural soil (IPCC, 2006). Based on the total N input tabulated with input-output analysis the N<sub>2</sub>O emissions were estimated with IPCC equation (Bouwman, 1996; Freney, 1997; IPCC, 2006; Smith *et al.*, 2001). In practice, the N<sub>2</sub>O was estimated on the basis of N input from chemical fertilizers, organic N from compost and manure and the amount of biological N fixation (BNF) in farm (Brown *et al.*, 2002; Mosier *et al.*, 1998). However, the modeling in the present study revealed that the significant amount of N enters the farm systems through water inflow of irrigation and rainfall. Two types of calculations were performed on the basis of two scenarios: (1) total N input in farm and (2) total N via water input (Table 4.14).

Table 4.14: N<sub>2</sub>O emissions from farms based on IPCC estimation

	Total N input in farm, kg N ha <sup>-1</sup> y <sup>-1</sup>		Total N via water input, kg N ha <sup>-1</sup> y <sup>-1</sup>	
	Total N	N <sub>2</sub> O emission	Total N	N <sub>2</sub> O emission
CF1	2009 (20) <sup>a</sup>	26.1 (1.2) <sup>a</sup>	109(12.7)	2.37(12.6)
CF2	1648 (11) <sup>a</sup>	21.6 (1.1) <sup>a</sup>	128(8.3)	2.60(1.10)
OF1	2718 (40) <sup>a</sup>	35.0 (1.5) <sup>a</sup>	406(6.1)	5.5(1.07)
OF2	1350 (140) <sup>a</sup>	17.9 (2.8) <sup>a</sup>	379(111)	5.7(2.38)

a SD: standard deviation (value in bracket)

The estimation of N<sub>2</sub>O emissions based on total N input in CF1, CF2, OF1 and OF2 indicated potential N flux of 26.1 ± 1.2, 21.6 ± 1.1, 35 ± 1.5 and 17.9 ± 2.8 kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively. Comparable results were reported which the annual cumulative N<sub>2</sub>O

emissions was  $29.2 \pm 3.7 \text{ kg N ha}^{-1} \text{ y}^{-1}$  in intensive vegetable farms with crop rotation of celery-choy sum-lettuce-bok choy (Jia *et al.*, 2012b). Similar results were also recorded by a seven years corn field study using static chamber in which the cumulative  $\text{N}_2\text{O}$  emissions range from 2.6 to  $78.8 \text{ kg N ha}^{-1} \text{ y}^{-1}$  ( $30$  to  $900 \text{ } \mu\text{g N m}^{-2} \text{ h}^{-1}$ ) (Ma *et al.*, 2010). Higher  $\text{N}_2\text{O}$  emissions in OF1 in comparison to other farms in this study might be due to the larger volume of composts input. This was supported by Jaiarree *et al.* (2014) who demonstrated the correlation of soil  $\text{N}_2\text{O}$  emissions and compost application rates. This study revealed the compost application rate of 30,000 and 50,000  $\text{kg ha}^{-1} \text{ y}^{-1}$  has resulted  $\text{N}_2\text{O}$  gaseous emissions of 2.56 and 3.47  $\text{kg N ha}^{-1} \text{ y}^{-1}$ , respectively and were significantly higher than non-fertilized soil ( $1.47 \text{ kg N ha}^{-1} \text{ y}^{-1}$ ).

The total N input of  $109 \pm 12.7$ ,  $128 \pm 8.3$ ,  $406 \pm 6.1$  and  $379 \pm 111 \text{ kg N ha}^{-1} \text{ y}^{-1}$ , respectively via irrigation and/or rainfall in CF1, CF2, OF1 and OF2 were estimated to contribute to  $2.37 \pm 12.6$ ,  $2.60 \pm 1.10$ ,  $5.5 \pm 1.07$  and  $5.7 \pm 2.38 \text{ kg N ha}^{-1} \text{ y}^{-1}$  of  $\text{N}_2\text{O}$  emissions, respectively. Research showed the  $\text{N}_2\text{O}$  emissions were directly related to the amount of mineral N available in the soil and increase do N fertilizer rates induced higher  $\text{N}_2\text{O}$  emissions (Hoben *et al.*, 2011; Li *et al.*, 2014; Rees *et al.*, 2013). Thus, the water flow is an important N input which emphasized the need to be included into the IPCC calculation to achieve a comprehensive modeling of N fluxes.

Microbial nitrification and denitrification process was the main source of  $\text{N}_2\text{O}$ , thus factors influencing this process will affect  $\text{N}_2\text{O}$  flux (Burford & Bremner, 1975; Butterbach-Bahl *et al.*, 2013). SOM and climate (rainfall and temperature) differences were identified as the primary factors of variability in  $\text{N}_2\text{O}$  emissions while soil tillage and nutrient regime were the most crucial farming practices that will induce soil  $\text{N}_2\text{O}$  emissions (Gao *et al.*, 2014). Soil C:N also plays an important role in N availability and



study indicated that soil C:N below 20:1-30:1 threshold has higher susceptibility to N mineralization which increased the chances of N leaching and N<sub>2</sub>O emissions (Haney *et al.*, 2012; Huang *et al.*, 2004). Other than that, types of crop and fertilizer, soil moisture, soil C content, soil pH and texture were factors that caused high variability of N<sub>2</sub>O emissions (Hénault *et al.*, 2012).

The estimation of N<sub>2</sub>O emissions based on IPCC equation was based on the total N input in farm systems and spatial variations were often ignored (Butterbach-Bahl *et al.*, 2013; Hénault *et al.*, 2012; Milne *et al.*, 2014). However, several authors suggested the correlation between an increased in N input and N<sub>2</sub>O emissions (Hoben *et al.*, 2011; Li *et al.*, 2014; Shcherbak *et al.*, 2014; Zhao *et al.*, 2015). This was supported by Rees *et al.* (2013) who highlighted the nutrient management has greater influence over N<sub>2</sub>O emissions as compared to spatial variation. Therefore, the IPCC estimation is still considered a low cost and effective method to estimate potential N<sub>2</sub>O emissions. Based on the IPCC estimation the global warming potential (GWP) of the total CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O emission for 20, 100 and 500 years from each study farms tabulated (Table 4.15). The results suggest that GWP in OF1 was the highest while CF1, CF2 and OF2 have similar GWP.

Table 4.15: The GWP of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O emission from study farms

	<b>Total GWP, kg CO<sub>2</sub> equivalents ha<sup>-1</sup></b>		
	20 years	100 years	500 years
CF1	13,594	13,620	10,696
CF2	13,835	13,687	11,206
OF1	78,561	78,596	74,676
OF2	14,313	14,130	12,051

## 4.9 Bacterial counts

The increased risk of contamination by enteric pathogens such as coliform, *E.coli*, and *Salmonella spp.* on vegetables due to organic nutrient source in farms have become a major concern on food safety (Oliver & Heathwaite, 2012; Sebesvari *et al.*, 2012). Study reported that 1.9% of farms that used non-organic nutrients and 9.7% of farms that used organic nutrients were tested positive for *E.coli* (Rosen & Allan, 2007). The objective of microbial plate count was to evaluate the level of microorganism presents in soils, pre-harvest vegetables and runoff water in each farm.

### 4.9.1 Aerobic plate count (APC)

The APC of soil samples in CF1, CF2, OF1 and OF2 was  $2.7 \times 10^6$ ,  $3.33 \times 10^6$ ,  $178 \times 10^6$  and  $17.5 \times 10^6$  CFU g<sup>-1</sup>, respectively (Figure 4.62). During the study period, the average soil APC in organic systems were significantly higher than conventional systems ( $P = 0.05$ ).

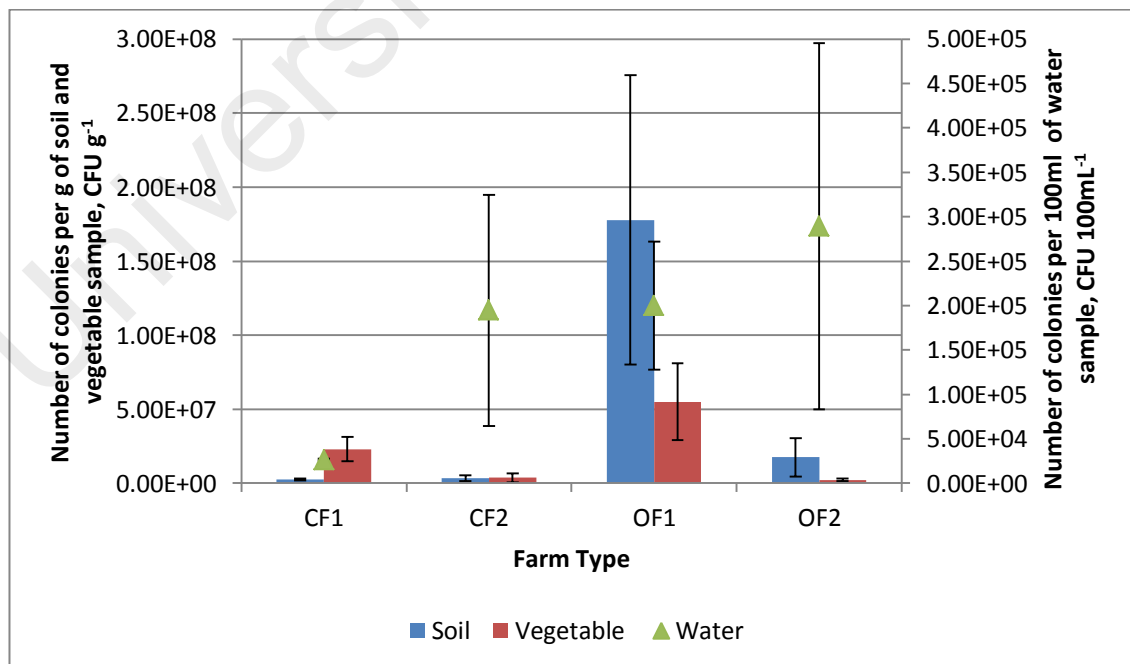


Figure 4.62: Average APC of soil, vegetable and water samples

However, no significant differences were observed in microbial population between each sampling ( $P = 0.46$ ) (Figure 4.63). Comparable results were reported that the soil microbial populations were about  $0.87 \times 10^8$  CFU  $g^{-1}$  under conventional onion farm and were significantly lower than organic onion farm that recorded  $7.87 \times 10^8$  CFU  $g^{-1}$  (Reilly *et al.*, 2013). Similarly, a study showed significantly higher APC in organic managed soil ( $57.8-78.8 \times 10^6$  CFU  $g^{-1}$ ) in comparison to non-organic managed soil ( $35.0-42.2 \times 10^6$  CFU  $g^{-1}$ ) (Bernard *et al.*, 2012). However, study by Meyer *et al.* (2015) suggested no significant differences of APC between organic (range from  $8.92 \times 10^6$  to  $1.26 \times 10^7$  CFU  $g^{-1}$ ) and conventional farms (range from  $3.40 \times 10^6$  to  $1.43 \times 10^7$  CFU  $g^{-1}$ ). However, the author noticed an increase of microbiological activity in organic farms as compared to conventional farms. This was supported by van Diepeningen *et al.* (2006) that higher biological activity,  $CO_2$  respiration and biodiversity was recorded in organically managed soils than the conventionally managed soils.

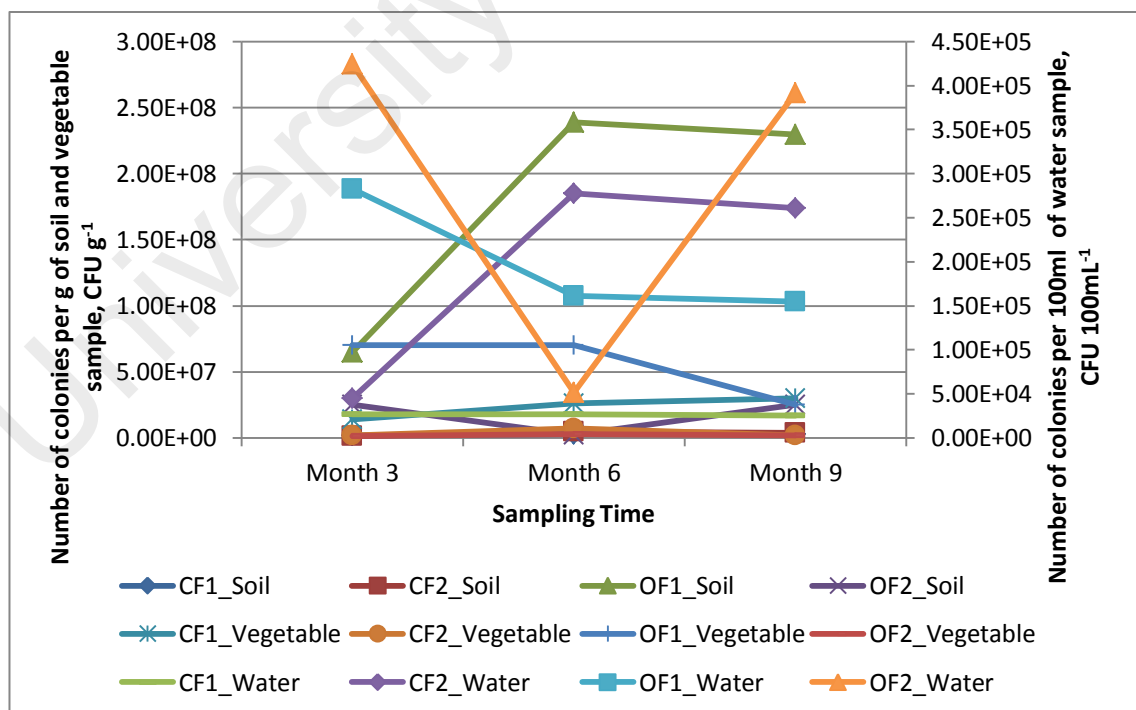


Figure 4.63: APC trend of soil, vegetable and water samples

Cultural practices such as organic amendments, rotations, tillage and use of biological control have significant and specific effects on soil microbial communities (Bernard *et al.*, 2012; Linn & Doran, 1984). In addition, soil properties such as salinity, microbial diversity, soil N and water soluble organic C were also affecting soil microbial populations (Ma *et al.*, 2013). Several authors reported that the supply of organic matters to soil through different amendments such as compost and manure stimulates microbial populations and increases microbial diversity due increased soil C availability (Chang *et al.*, 2007; Chang *et al.*, 2008; Fraser *et al.*, 1988; Gomez *et al.*, 2006; Reilly *et al.*, 2013; Tao *et al.*, 2015; van Diepeningen *et al.*, 2006). This was supported by a study that reported compost amended soil contained 20-60% higher bacteria counts as compared to soil without compost (Bernard *et al.*, 2012). Thus, compost used in the organic systems in this study might induce higher populations of soil bacteria. The higher soil microbial populations in OF1 as compared to OF2 might be due to the variations in compost sources and application rates. The Bokashi compost used in OF1 contained effective microbes that might induce soil microbial population. This was supported by Chang *et al.* (2008) who suggested that different types of organic fertilizer have various effects on soil microbial. The microbial population in CF2 was slightly higher as compared to CF1 due to the high manure application. This was supported by Chang *et al.* (2007) that the increased of manure application in soils have greater microbial count and populations. Study also showed that soil microbial population increased with organic fertilizer application but ceased at optimal point (Chang *et al.*, 2007). Besides organic fertilizer, the water used for irrigation was a critical sources of microbial contamination in farms (Zaman *et al.*, 2014).

Microbes can easily transmit from soil to vegetable via transporting vector such as water splash, human and animal. The APC of pre-harvest vegetable samples in CF1,

CF2, OF1 and OF2 were  $23 \times 10^6$ ,  $3.71 \times 10^6$ ,  $55 \times 10^6$  and  $2.22 \times 10^6$  CFU g<sup>-1</sup>, respectively. Unlike soil APC no significant difference was observed in vegetable APC between organic and conventional systems. Similar results reported that in organic herbs farms cultivating fresh mint, jasmine, and lemongrass has average APC of  $0.03 \times 10^6$ ,  $0.5 \times 10^6$ , and  $0.7 \times 10^6$  CFU g<sup>-1</sup> ( $4.5 \pm 0.12$ ,  $5.7 \pm 0.11$ ,  $5.9 \pm 0.11$  log CFU g<sup>-1</sup>), respectively (Zaman *et al.*, 2014). In addition, study in leafy vegetable farms also reported similar APC ranged between  $0.1 \times 10^3$  and  $1 \times 10^6$  CFU g<sup>-1</sup> (2 log and 6 log CFU g<sup>-1</sup>) (Cardamone *et al.*, 2015). Besides from soil, irrigation was also one of the major contributors to microbial contamination on vegetable. This was supported by a long-term experiment on vegetables under sewage irrigation whereby the vegetable irrigated with contaminated water contained microbial population ranged between  $2 \times 10^6$  and  $3.5 \times 10^7$  CFU g<sup>-1</sup> (Minhas *et al.*, 2006). The author highlighted that the use of sewage as irrigation water increased microbial population on vegetable produce.

Runoff from farms may wash away soil particles along with microbes from agricultural soil. The APC of runoff water samples in CF1, CF2, OF1 and OF2 was  $0.27 \times 10^5$ ,  $1.95 \times 10^5$ ,  $2.0 \times 10^5$  and  $2.89 \times 10^5$  CFU 100mL<sup>-1</sup>, respectively. No significant difference in APC between each farm ( $P = 0.162$ ), between organic and conventional systems ( $P = 0.117$ ) and between each sampling time ( $P = 0.706$ ). Factors such as precipitation intensity and volume, slope of land, climate, vegetation, soil properties, soil nutrients availability, wildlife, organic fertilizer types and application rates, fertilizer application method affect runoff APC (Jamieson *et al.*, 2002; Patni *et al.*, 1985). Thus, the lack of significant difference in runoff APC between farm systems indicated low variation between farms or lack of influence by these factors. Study suggested potential risk of faecal bacterial transport via runoff to surface water may lead to faecal contamination of recreational and drinking water sources (Jamieson *et al.*, 2002).

#### 4.9.2 Coliform/*E.coli* count (CE)

The coliform/*E.coli* count of soil samples in CF1, CF2, OF1 and OF2 was  $32 \times 10^3$ ,  $1.38 \times 10^3$ ,  $81.8 \times 10^3$  and  $5.72 \times 10^3$  CFU g<sup>-1</sup>, respectively (Figure 4.64). There was significant difference of coliform/*E.coli* count between each farm ( $P = 0.036$ ). However, no significant difference was observed between organic and conventional systems ( $P = 0.284$ ) and between each sampling ( $P = 0.476$ ) (Figure 4.65). Comparable results were reported by Zaman et al. (2014) where organic farms cultivating mint, jasmine and lemongrass contained  $15 \times 10^3$ ,  $126 \times 10^3$  and  $631 \times 10^3$  CFU g<sup>-1</sup> (4.2, 5.1, and 5.8 log CFU g<sup>-1</sup>) of total *E.coli* count. Lower level of *E. coli* were detected in soil samples from surveyed strawberry farms which range from 1 to 3.3 x log CFU g<sup>-1</sup> ( $0.1 \times 10^2$  -  $1.9 \times 10^3$  CFU g<sup>-1</sup>) (Johannessen et al., 2015). Manure and irrigation water were the probable sources of *E.coli* (Islam et al., 2004; VanderZaag et al., 2010). This was supported by Wood (2013) and Reynnells et al. (2014) that demonstrated water used for irrigating the fields or surface water runoff can disseminate pathogen around farming areas. Both CF1 and CF2 applied chicken manure as soil amendments thus high possibilities the source of *E.coli* is from the livestock manure. Livestock manure was not used in organic systems OF1 and OF2 thus the potential sources of coliform/*E.coli* might be from the compost application and irrigation. Study suggested incomplete thermal inactivation during composting or improper aged compost allowed *E.coli* to survive and regrow in finished product (Ackers et al., 1998; Reynnells et al., 2014). In addition, study also showed potential contamination of compost by fresh manure due to improper handling of finished compost (Zaman et al., 2014).

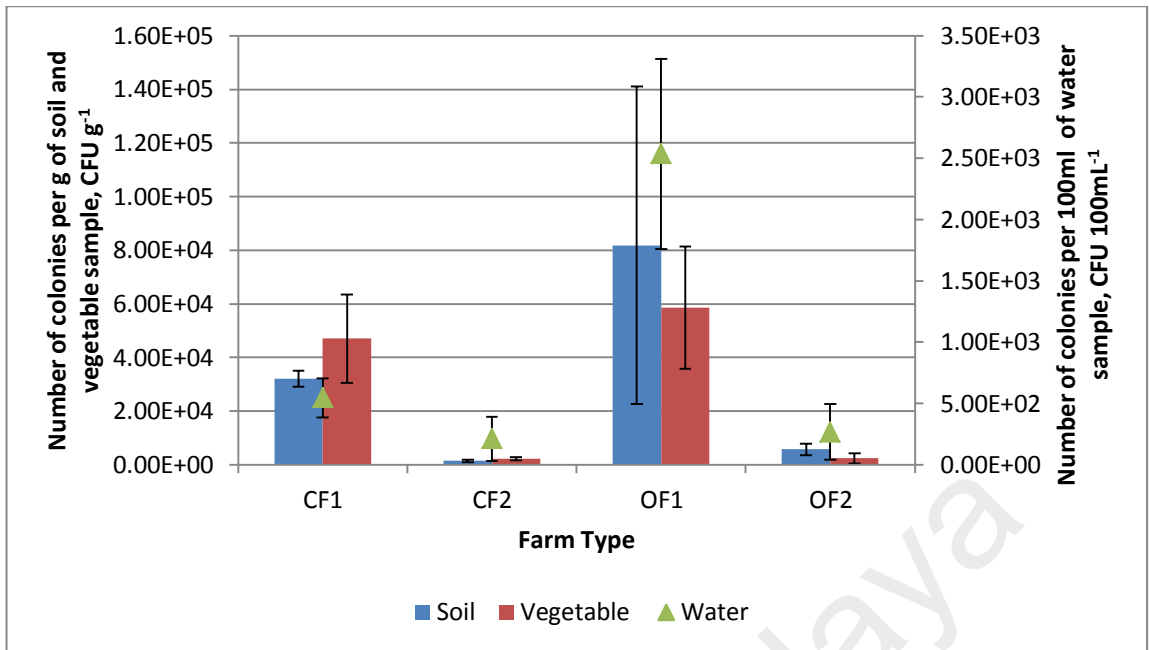


Figure 4.64: Aerobic coliform/*E. coli* count of soil, vegetable and water samples

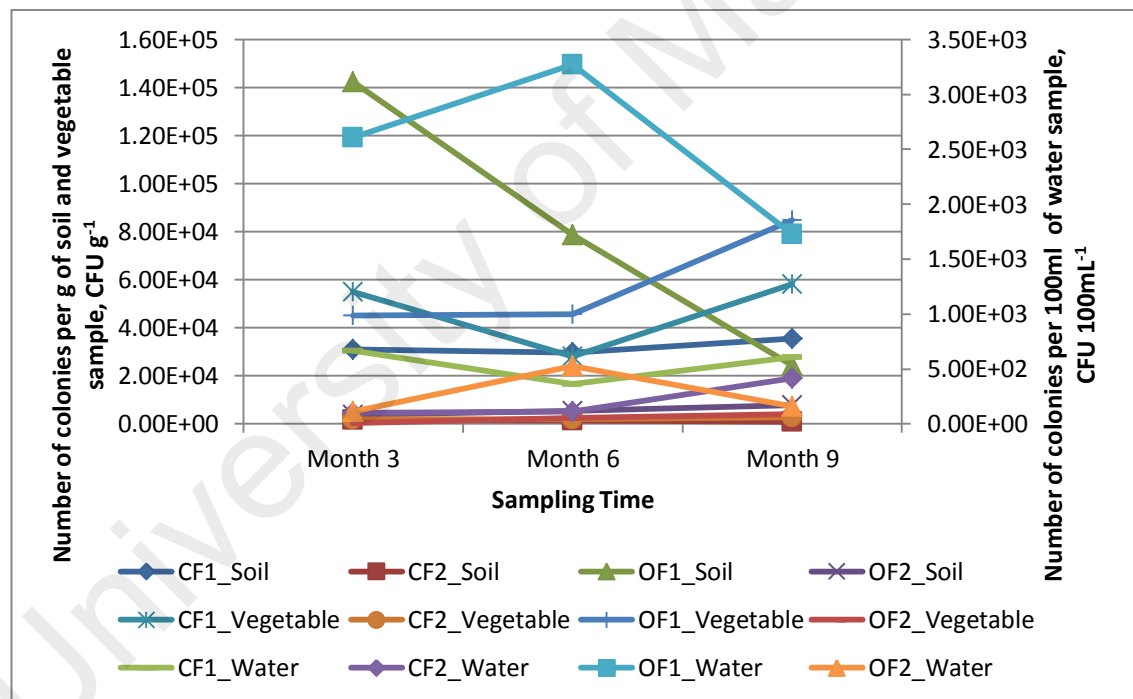


Figure 4.65: Trend of aerobic coliform/*E. coli* count of soil, vegetable and water samples

In addition, a statistical analysis revealed that the compost C:N ratio, total organic C, and moisture content influenced the pathogen regrowth (Reynnells *et al.*, 2014). However, some suggested that the consistent existence of *E. coli* in the field might be from the naturalized strains of *E. coli* that become part of the indigenous microflora of

soils, sand, sediments, and algae in tropical, subtropical, and temperate environments (Ishii & Sadowsky, 2008; Ishii *et al.*, 2010; VanderZaag *et al.*, 2010)

Soil is one of the key source of coliform/*E.coli* contamination to vegetables (Liu *et al.*, 2013a). Study showed that the leafy vegetables have higher bacterial count as compared to other vegetable products and it is likely associated with broad and rough surface morphology of leaf (Cardamone *et al.*, 2015). The soil coliform/*E.coli* count indicated presence of pathogen which might lead to contamination on vegetables. The coliform/*E.coli* count on pre-harvest vegetable samples in CF1, CF2, OF1 and OF2 were  $47.1 \times 10^3$ ,  $2.21 \times 10^3$ ,  $58.6 \times 10^3$  and  $2.36 \times 10^3$  CFU g<sup>-1</sup>, respectively. There were significant differences of coliform/*E.coli* count between each farm (P = 0.002). Similar results were reported in organic farm cultivating fresh mint, jasmine, and lemongrass with the total *E.coli* count of  $1.5 \times 10^4$ ,  $12 \times 10^4$ ,  $60 \times 10^4$  CFU g<sup>-1</sup> (4.2, 5.1, 5.8 log CFU g<sup>-1</sup>), respectively (Zaman *et al.*, 2014). In addition, study in leafy vegetables farms indicated present of *E.coli* about  $0.1 \times 10^3$  -  $1 \times 10^3$  CFU g<sup>-1</sup> on salad and spinach (Cardamone *et al.*, 2015). Similarly, the total coliform count on pre-harvest lettuce sample in traditional, organic and hydroponic farm were  $> 2.4 \times 10^3$ ,  $> 2.4 \times 10^3$ , and  $0.6 \times 10^2$  CFU g<sup>-1</sup> (Gomes Neto *et al.*, 2012). Greater coliform/*E.coli* on vegetables in CF1 and OF1 might be due to higher soil coliform/*E.coli* in these two farms. The vegetable coliform/*E.coli* count in all the farms in this study were classified as “good” according to HACCP guidelines in which food containing microorganism more than  $5 \times 10^7$  CFU g<sup>-1</sup> were classified as “spoiled” (Minhas *et al.*, 2006). Study revealed washing of vegetable reduced pathogen levels on vegetables (Natvig *et al.*, 2002). Thus, the washing of vegetable postharvest will further reduce the coliform/*E.coli* count on vegetable produce in the farms in this study.



The variation of irrigation sources, types of compost and manure, and crop types in the study farms lead to different *E.coli* count. A long-term experiment of vegetable farms suggested the source of irrigation determined the population of *E.coli* on vegetable because the direct contact of water onto vegetables (Minhas *et al.*, 2006). This was supported by Safi *et al.* (2011b) who highlighted the *E.coli* count varied for different sources of irrigation water.

Coliform/*E.coli* was found to be present in soils of all the farms in this study. Thus there was a high possibility that the runoff from the farms in this study also contained coliform/*E.coli*. The coliform/*E.coli* count of runoff water samples in CF1, CF2, OF1 and OF2 indicated total of  $5.45 \times 10^2$ ,  $2.11 \times 10^2$ ,  $25.4 \times 10^2$  and  $2.67 \times 10^2$  CFU 100 mL<sup>-1</sup> of coliform/*E.coli* load exit the farm systems via runoff, respectively. There was a significant difference of coliform/*E.coli* count between each farm ( $P = 0.0004$ ). Comparable results were reported where the *E. coli* concentrations in runoff range from  $1.9 \times 10^3$  to  $2.8 \times 10^4$  (Mishra *et al.*, 2008). A farm simulation demonstrated higher coliform count of  $1.5 \times 10^3$ ,  $2.4 \times 10^5$  and  $1.8 \times 10^6$  CFU 100 mL<sup>-1</sup> in control, partial conventional and rotational grazing plot (Edwards *et al.*, 2000). Different results reported might be due to the variation and influence of precipitation intensity and volume, slope of land, climate, vegetation, soil properties, soil nutrients availability, wildlife, organic fertilizer types and application rates, fertilizer application method (Jamieson *et al.*, 2002; Patni *et al.*, 1985).

The coliform/*E.coli* count of runoff in farms in this study indicated pathogens were being washed off from the farm via runoff. This was supported by Liu *et al.* (2013) which identified surface runoff as one of the transmissions vehicle for *E. coli* in agriculture area. Study also showed high survival ability of *E. coli* in runoff which can

remain in high population even after 46 days of manure application (McDowell *et al.*, 2006). Thus, *E.coli* was often used as indicator for faecal contamination in waterways (Ishii & Sadowsky, 2008). The runoff *E.coli* count in all farms in this study were below the WHO limit  $<1000 \text{ CFU } 100 \text{ mL}^{-1}$  for faecal coliform in irrigation water (Minhas *et al.*, 2006; Safi *et al.*, 2011b). According to European Bathing Water Legislation for all water type (76/160/EEC), the coliform/*E.coli* level in runoff of CF1 falls in the category of “Good Quality” ( $<10,000 \text{ CFU } 100 \text{ mL}^{-1}$ ) while CF2, OF1 and OF2 are characterized as “Excellent Quality” ( $<500 \text{ CFU } 100 \text{ mL}^{-1}$ ) (Mansilha *et al.*, 2010).

#### **4.9.3 *Salmonella/Shigella* count (SS)**

*Salmonella/Shigella* (SS) count of soil samples in CF1, CF2, OF1 and OF2 was  $2.7 \times 10^4$ ,  $0.64 \times 10^4$ ,  $45 \times 10^4$  and  $0.88 \times 10^4 \text{ CFU g}^{-1}$ , respectively (Figure 4.66). There was significant difference of SS count between each farms ( $P = 0.001$ ). However, no significant differences were observed between organic and conventional systems ( $P = 0.079$ ) and also between each samplings ( $P = 0.453$ ) (Figure 4.67). Comparable results were reported by Zaman *et al.* (2014) in which the soil from organic herbs farms with manure application contained  $0.1 \times 10^6$  ( $5.0 \log \text{ CFU g}^{-1}$ ) of total *Salmonella Shigella* (SS) count. The SS count indicated potential SS contamination in soil of all the farm systems. The source of SS in farms in this study might be from livestock manure, contaminated compost and irrigation water, and wild or domestic animals (Hanning *et al.*, 2009). The SS count of pre-harvest vegetable samples in CF1, CF2, OF1 and OF2 was  $3.86 \times 10^4$ ,  $1.02 \times 10^4$ ,  $4.22 \times 10^4$  and  $0.15 \times 10^4 \text{ CFU g}^{-1}$ , respectively. There were significant differences of SS count between each farm ( $P = 0.014$ ). This is supported by Cardamone *et al.* (2015) who similar results of  $9.5 \times 10^6 \text{ CFU g}^{-1}$  ( $6.98 \log \text{ CFU g}^{-1}$ ) *Salmonella spp.* in green salad (Cardamone *et al.*, 2015).

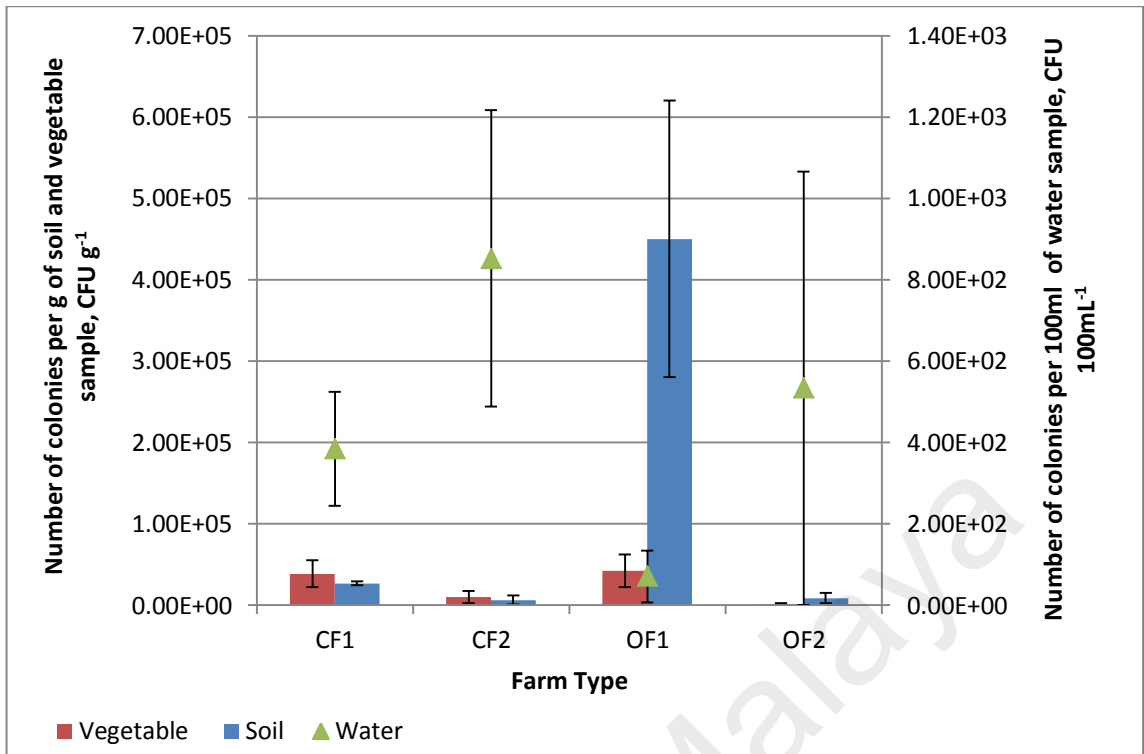


Figure 4.66: Aerobic SS count of soil, vegetable and water samples

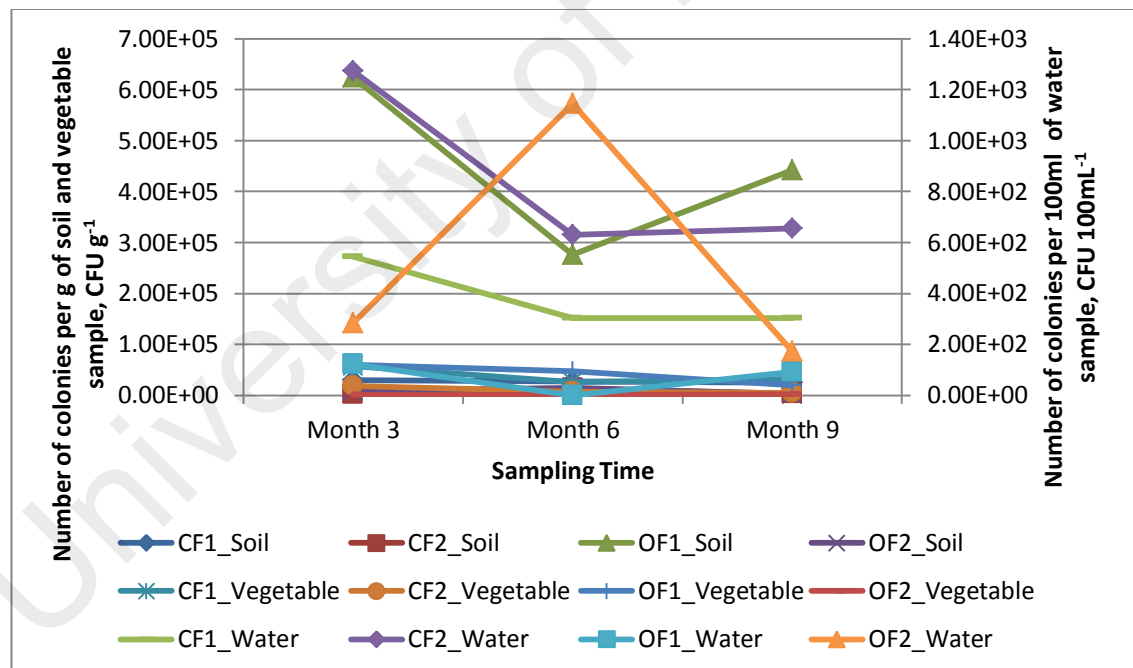


Figure 4.67: Trend of aerobic SS count of soil, vegetable and water samples

In addition, *Salmonella spp.* also found in lettuce samples in Ouagadougou, Burkina Faso with 50% prevalence (Traoré *et al.*, 2015). According to Pan *et al.* (2015) organic farming practices have doubled the probability of vegetable contaminated with

*Salmonella spp.*, however, no significant differences of SS count were observed between organic and conventional systems ( $P = 0.85$ ) in this study. The variation of SS level in different study might be due to methodologies of SS count and farm management.

There is a high possibility that SS contamination on vegetables were from the soils as previous section indicated presence of SS in all the farm soils. Other than that, manure, soil, surface water, sewage and wildlife were likely to be the contamination sources on pre-harvest vegetable (Liu *et al.*, 2013a). This was supported by Islam *et al.* (2004b) that highlighted the important roles of contaminated manure, compost and irrigation on pathogen contamination in soils and vegetables. *Salmonella spp.* contamination increased parallel with *E.coli* bacteria count in which increment of every 100 CFU g<sup>-1</sup> of *E.coli* is pair with 15-30% increased of SS contamination (Pan *et al.*, 2015). Similar results observed in this study whereby both CF1 and OF1 have higher *E.coli* and SS in soil and water samples. The vegetable SS count in all the farms in this study were classified as “good” according to HACCP guidelines (Minhas *et al.*, 2006). The washing of vegetable in postharvest stage in the farms in this study might reduce pathogen levels on vegetables (Natvig *et al.*, 2002).

Several studies suggested that the manure application significantly increased bacterial loading to downstream water bodies especially when rainfall or irrigation occurs right after manure application (Frey *et al.*, 2015; Mishra *et al.*, 2008; VanderZaag *et al.*, 2010). In addition, a study highlighted the common presence of *Salmonella spp.* in aquatic systems (Polo *et al.*, 1999; Traoré *et al.*, 2015). Thus, it is crucial to evaluate the SS count in runoff from each farm to assess bacterial load and identified potential contamination. The SS count of runoff water samples in CF1, CF2, OF1 and OF2

showed  $3.84 \times 10^2$ ,  $8.53 \times 10^2$ ,  $0.71 \times 10^2$  and  $5.34 \times 10^2$  CFU  $100 \text{ mL}^{-1}$  of SS load exited the farm systems. This SS count in runoff water indicated the runoff washed off SS from the farm soil. This was supported by Sigua et al., (2010) which 33% to 67% of agriculture runoff under various management systems contained *Salmonella spp.* In addition, Jacobsen & Bech (2012) also suggest the water runoff transported organic matter such as manure that contained *Salmonella spp.*

Different SS count in each farm might be due to variations of pathogen transfer rate via runoff. Research showed that transfer of pathogen via runoff depends on water flow, soil particle size, soil electric charge and soil hydrophobicity (Jacobsen & Bech, 2012). The runoff SS count in all the farms in this study were below WHO standards for irrigation water ( $<1000$  CFU  $100 \text{ mL}^{-1}$  of SS) (Safi et al., 2011b). However, the *Salmonella spp.* in water runoff of the farms in this study had exceeded the recommended quality for bathing water ( $100$  CFU  $100 \text{ mL}^{-1}$ ) according to European Bathing Water Legislation for all water type (76/160/EEC) (Mansilha et al., 2010)

Based on the field observation, flies were commonly found in all the farms regardless of organic or conventional systems. Flies are the usual vector for transmitting pathogen from farm to farm. This was supported by Adebayo-Tayo et al. (2012) that reported flies can harbour pathogen which the total bacterial, coliform and SS counts were range from  $4.8 \times 10^4$  to  $18.9 \times 10^4$  CFU  $\text{g}^{-1}$ ,  $4.9 \times 10^4$  to  $13.0 \times 10^4$  CFU  $\text{g}^{-1}$ , and  $4.1 \times 10^4$  to  $2.16 \times 10^4$  CFU  $\text{g}^{-1}$ , in CF1, CF2, OF1 and OF2, respectively. According to the farmers of CF1 and CF2, the flies were coming from nearby oil palm plantation which applies manure to soil as amendment. Thus, there is a high possibility of pathogen transmission between farms. It was reported the use of livestock manure to fertilize crop increased the risk of *E. coli* contamination in organic farm (Bourn & Prescott, 2002; Mukherjee et

*al.*, 2007). Thus, the used on manure in CF1 and CF2 might be one of the contamination source. In addition, organic waste and manure were identified as key sources of microbial contamination in agricultural land (Bonti-Ankomah *et al.*, 2006; Bourn & Prescott, 2002; IFST, 2013; Santamaría & Toranzos, 2003; Yiridoe *et al.*, 2005). In OF1 and OF2 manure was not used a soil amendments. However, the pathogen count suggested the composts used in OF1 and OF2 might be improper aged due to incomplete thermal inactivation or contaminated due to improper handling and storage which allowed pathogen to survive and regrow (Ackers *et al.*, 1998; Reynnells *et al.*, 2014; Zaman *et al.*, 2014).

The microbial count of soil, vegetable and runoff water samples indicated potential of *E.coli* and SS contamination in all the farms in this study. This poses questions of food safety and whether farm produces are exposed to higher level of microbial contaminant. Agriculture soil contains enteric pathogens mostly due to use of manures and biosolids. The common enteric pathogens found in agriculture soil are *E.coli*, *Salmonella spp.*, and *Shigella spp.* which have the ability to persist in the soil for 8 to 12 weeks after fertilizing (Johannessen *et al.*, 2005). Pathogen in soils can spread by runoff and contaminates neighboring soil and water bodies (Santamaría & Toranzos, 2003). Thus, the used of recycled water as irrigation in the study farms increases the chances of spreading pathogen around the farms areas (Section 4.7). The use of contaminated irrigation water increased the chances of pathogen contamination on vegetable produce (Oliveira *et al.*, 2010). The pathogen in runoff and leachate can cycle back to farm or even affecting neighboring farms if the contaminated water is used for irrigation purpose (Bourn & Prescott, 2002). The vegetables were easily get contact with soil even after harvest which can lead to contamination if the soil contains pathogen (Guo *et al.*, 2002; Santamaría & Toranzos, 2003). Therefore, it is suggested to all the farms in this

study to filter the runoff before recycled back into the farms in order to avoid cross contamination within the farming areas.

There were several limitations of bacterial counts in this study: First, the bacterial count provides an estimation of potential bacterial count present in soil, pre-harvest and runoff water. However, the estimation do not reflect the prevalence of pathogen in each sample as the bacterial count was based on composite samples (n=25). This is especially true for vegetable samples as not every load of vegetables was contaminated by coliform/*E.coli* or SS. Study of 484 leafy green farm indicated overall prevalence of *E. coli* was only 0.7% (Wood, 2013). Second, the coliform/*E.coli* and SS count estimated the coliform/*E.coli* and *Salmonella/Shigella* as a whole. Thus, it provides only a rough estimation of the potential coliform/*E.coli* and SS presents in the samples. Third, the bacterial count for vegetables in this study was only on pre-harvest vegetable. Thus, minimal processing in postharvest stage such as washing will further reduce the microbial population on vegetables.

#### **4.10 Farm management efficiency**

##### **4.10.1 Yield**

Farm productions in each farms in this study varies significantly ( $P < 0.01$ ) with the average leafy vegetable yield of  $610.8 \pm 342$ ,  $108 \pm 12$ ,  $12.6 \pm 5.3$  and  $203.9 \pm 156 \text{ t ha}^{-1} \text{ y}^{-1}$  in CF1, CF2, OF1 and OF2, respectively. The average yield in conventional systems was significantly higher than conventional farm systems ( $P = 0.002$ ). The monthly productions in CF2 and OF2 remain steadily throughout the year except for Choy Sum productions in OF2 that peak in September (Figures 4.68 and 4.69).

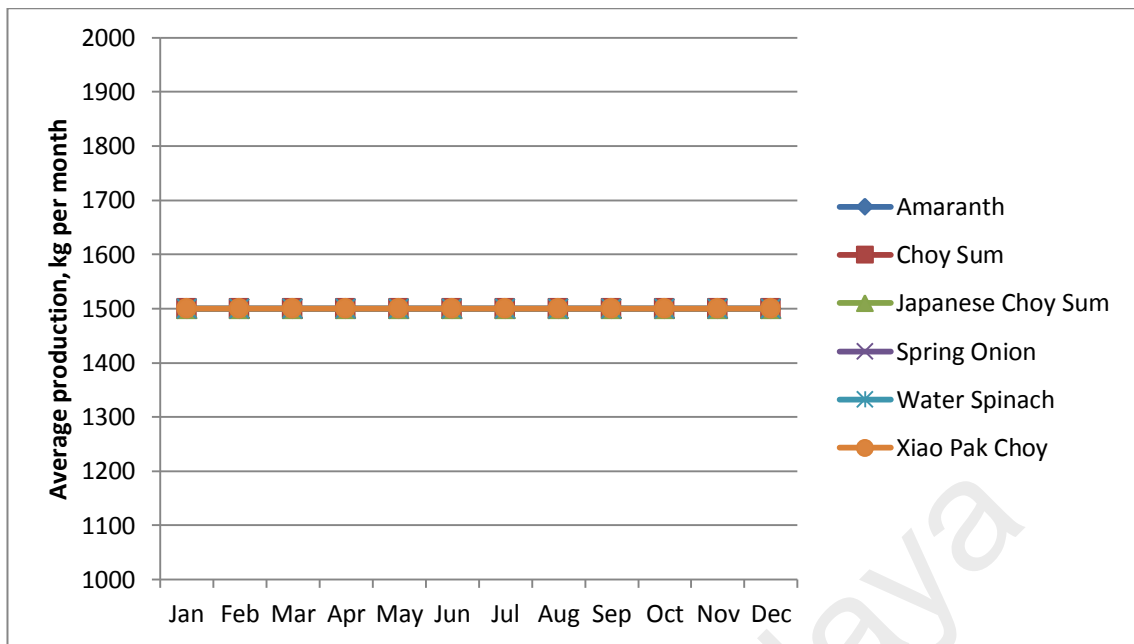


Figure 4.68: Average monthly production of leafy vegetables in CF2

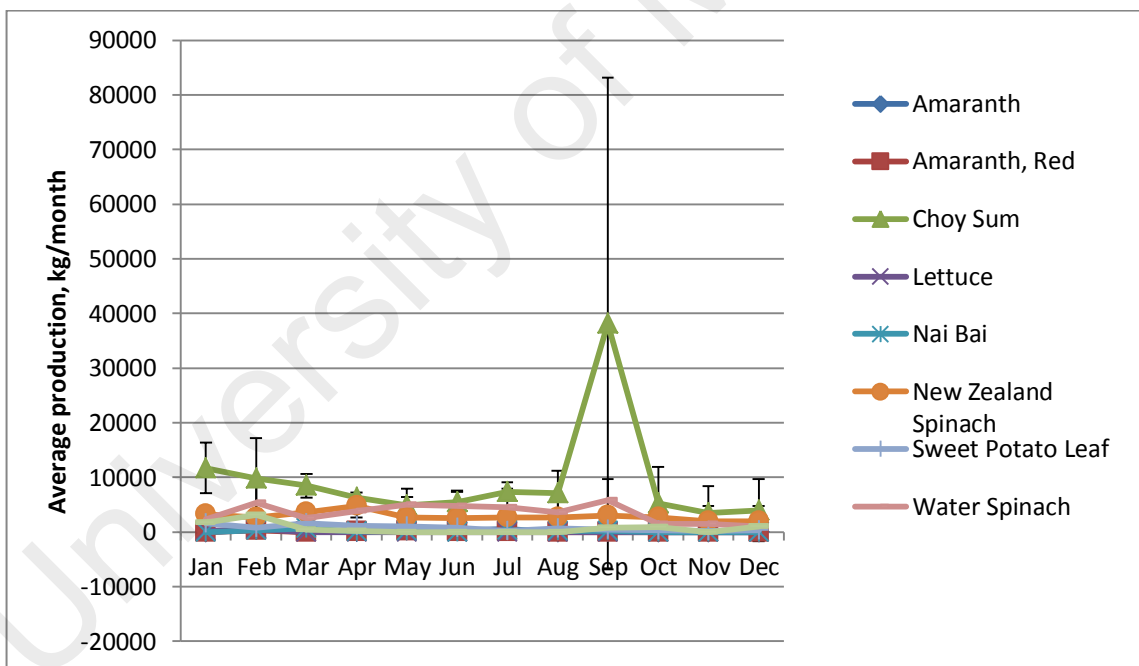


Figure 4.69: Average monthly production of leafy vegetables in OF2

In comparison, the monthly productions in CF1 increased steadily over the year (Figure 4.70). In OF1, the vegetable productions fluctuated drastically for most variety of leafy vegetable (Figure 4.71). Several authors also reported lower yield in organic farms as compared to conventional farms (de Ponti *et al.*, 2012; Patil *et al.*, 2014; Ponisio *et al.*,



2015; Rosen & Allan, 2007; Seufert *et al.*, 2012; Tuomisto *et al.*, 2012). The lower yield from organic systems indicated more land was required to achieve equivalent produce to conventional systems (Tuomisto *et al.*, 2012). This might lead to deforestation and affect biodiversity and thus undermining the environmental benefits of organic practice. Interestingly, the vegetable yield in OF2 ( $203.9 \pm 156 \text{ t ha}^{-1} \text{ y}^{-1}$ ) was higher than CF2 ( $108 \pm 12 \text{ t ha}^{-1} \text{ y}^{-1}$ ) which suggested with appropriate farm management organic farm can achieve comparable yield or even higher than conventional farm. A study reported that organic systems were capable to achieve equal or even higher yields than conventional farms especially in developing countries (Scialabba & Müller-Lindenlauf, 2010). This was supported by Seufert *et al.* (2012) and Ponisio *et al.* (2015) who suggested under particular condition, crop types, growing condition coupled with good management practices and agro-ecological farming approach allowed organic systems to match conventional yields. This highlighted the importance of farm management in order to achieve target yield.

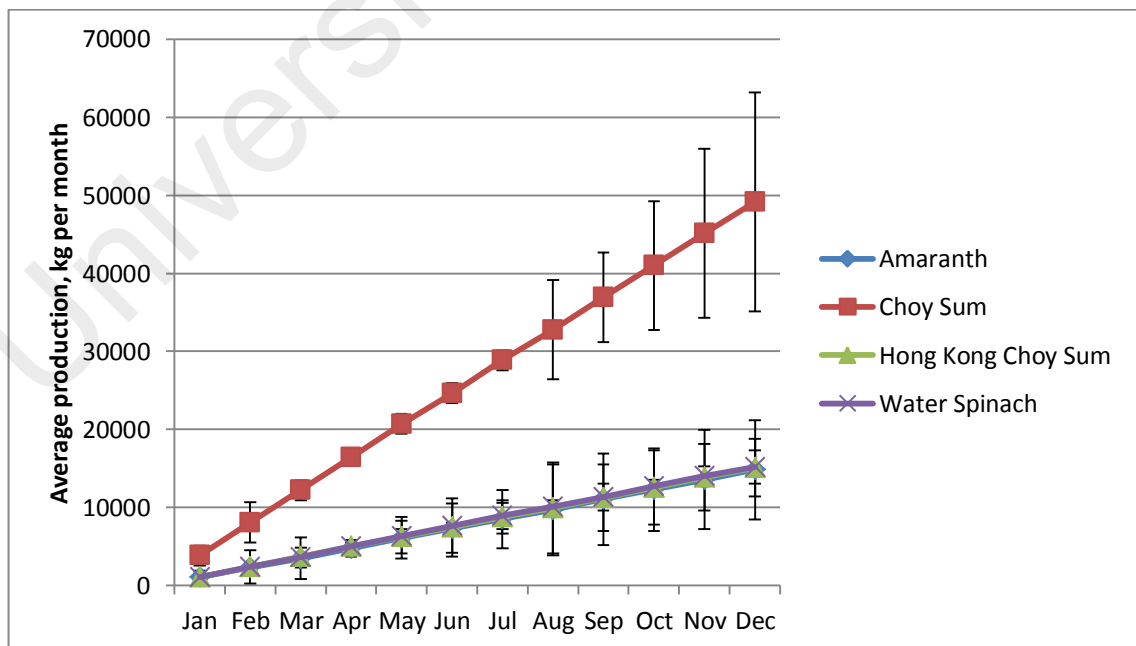


Figure 4.70: Average monthly production of leafy vegetables in CF1

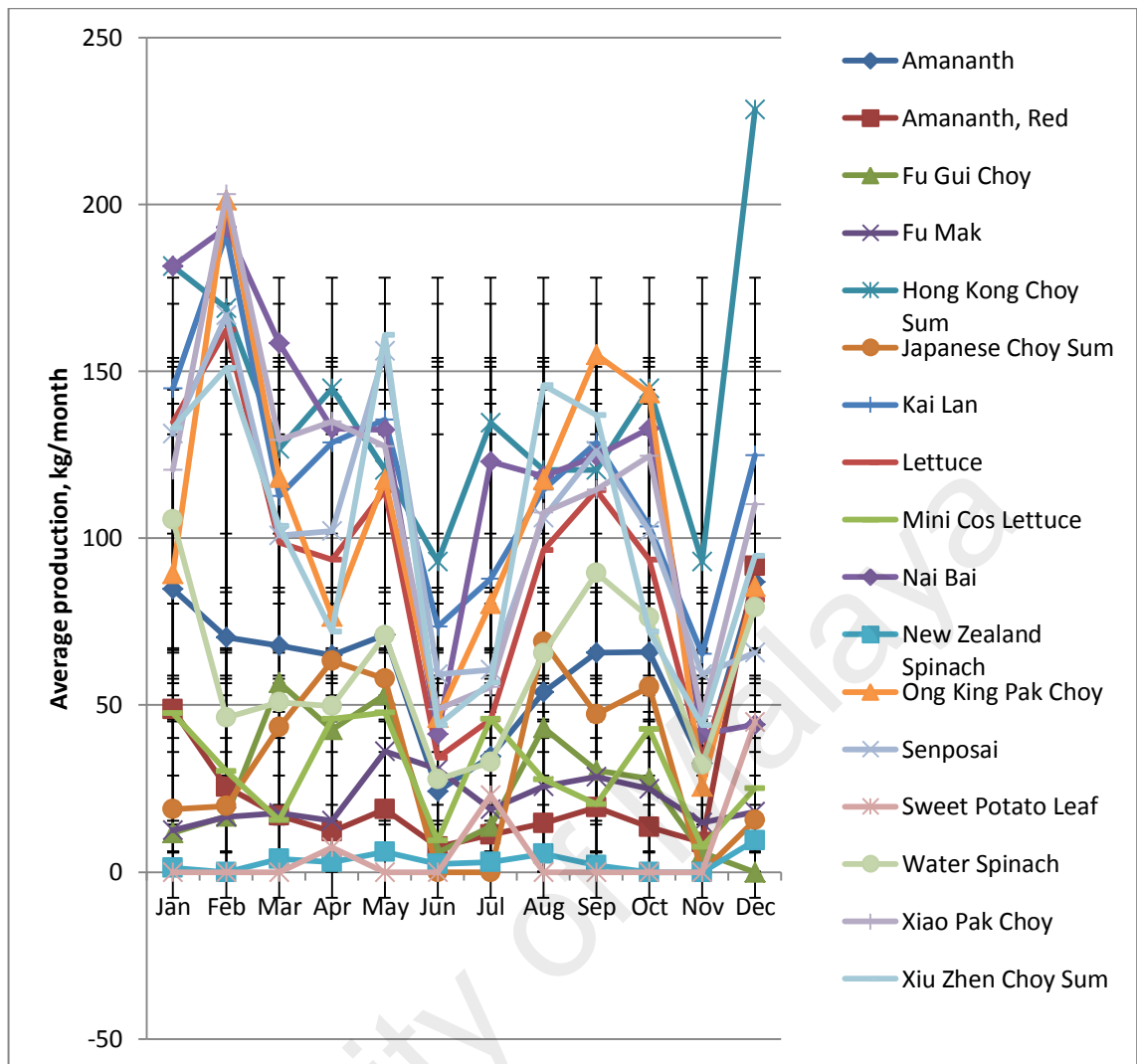


Figure 4.71: Average monthly production of leafy vegetables in OF1

#### 4.10.2 Water use efficiency on productivity (WUEP)

High water usage for agriculture activities has been a concerned especially under the pressure on renewable water resources for irrigation and water scarcity (FAO, 2009, 2015a; UNESCAP, 2014). In general the WUEP in the farms in this study was highest in OF2 with  $7 \text{ kg m}^{-3}$  followed by OF1 ( $2 \text{ kg m}^{-3}$ ), CF1 ( $1 \text{ kg m}^{-3}$ ), and CF2 ( $0.03 \text{ kg m}^{-3}$ ) (Table 4.16). A four years study in drip irrigated cabbage field showed comparable WUEP results to OF2 which is about  $7.8 \text{ kg m}^{-3}$  (Al-Said *et al.*, 2012). Results similar to OF1, CF1 and CF2 were reported by Scheierling *et al.* (2014) that WUEP for wheat, rice and maize ranged  $0.6\text{-}1.7$ ,  $0.6\text{-}1.6$ , and  $1.1\text{-}2.7 \text{ kg m}^{-3}$ , respectively.

Table 4.16: WUEP, synthetic fertilizer use efficiency, manure use efficiency, and compost use efficiency

	CF1	CF2	OF1	OF2
WUEP, kg m <sup>-3</sup>	1	0.03	2	7
Synthetic fertilizer use efficiency, kg kg <sup>-1</sup>	2	2	NA	NA
Manure use efficiency, kg kg <sup>-1</sup>	13	0.19	NA	NA
Compost use efficiency, kg kg <sup>-1</sup>	15	NA	0.15	3

NA: not available

Different WUEP reported might be due to the complex interrelation between vegetation, agronomy, and engineering (De Pascale *et al.*, 2011). Lower WUEP in conventional farm CF1 and CF2 might be due to the that variations in irrigation systems. The main irrigation in CF1 and CF2 were rainfed and sprinkler while OF1 and OF2 are solely depends on sprinkler irrigation. Study highlighted variation in irrigation systems can lead to different water use efficiencies in farms (Qadir *et al.*, 2010; Zotarelli *et al.*, 2009), eg. sprinkler irrigation increased the irrigated area by 20% to 30% as compared with furrow irrigation (De Pascale *et al.*, 2011). The key to enhance farm WUEP was to increase produce per unit of water consumed, reduce water loss to sinks, degradation of water quality and practice of water recycling (Howell, 2001; Scheierling *et al.*, 2014). Thus, selection of irrigation method, irrigation schedule, water recycling, and conservation agriculture are crucial tool in efficient water management (Jensen *et al.*, 2014).

#### 4.10.3 N use efficiency on productivity (NUEP)

The synthetic fertilizer use efficiency of productivity is about 2 kg produce per kg fertilizer for both CF1 and CF2. However, the manure use efficiency of productivity was higher in CF1 (13 kg produce per kg manure) in comparison to CF2 which was about 0.19 kg produce per kg manure. The compost use efficiency of productivity in CF1, OF1 and OF2 were about 15, 0.15 and 3 kg of produce per kg of compost. The

synthetic fertilizer, manure and compost efficiency provided a basic nutrient use efficiency in terms of soil amendment quantity, thus for better understanding of N use in the farms the N use efficiency of production (NUEP) were calculated (Table 4.17). Based on total vegetable N and total N input in farm, the NUEP in OF2 recorded the highest (27.6%) followed by CF1, OF1 and CF2 which were about 17.7, 4.2 and 2.3%, respectively. Comparable results were observed in a study of crop production in 31 province of China where the NUEP ranged from 12% to 45% in year 2005 (Ma *et al.*, 2012). Similar NUEP range from 12-33% were reported for intensive vegetable rotations in tropical Andisols (volcanic soil) (Widowati *et al.*, 2011). The NUEP was affected by several factors such as soil properties (N, P, K and C content), nutrient reserve, crops efficiency, climate, fertilizer type (N source, rate, application method), mycorrhiza, nutrient dynamic of soil amendments (lime, manure, and compost), tillage and irrigation (Abalos *et al.*, 2014; Baligar *et al.*, 2001; Chen *et al.*, 2004). Thus, the complex relationship between all these factors caused the NUEP varies between the each study farm.

Table 4.17: NUEP in farming systems

	CF1	CF2	OF1	OF2
Total vegetable production, kg ha <sup>-1</sup> yr <sup>-1</sup>	355	38	114	372
Total N input , kg ha <sup>-1</sup> yr <sup>-1</sup>	2009	1648	2718	1350
Total N loss, kg ha <sup>-1</sup> yr <sup>-1</sup>	1014	1002	1555	1211
N use efficiency fraction, %	17.7	2.3	4.2	27.6
N loss fraction	3	27	14	3

The low NUEP in CF2 and OF1 indicated that the soil available N might be loss from farm systems without being utilized for crop growth. Organic systems were presumably more efficient in nutrient management due to lower nutrient use (Rosen & Allan, 2007; Tuomisto *et al.*, 2012). However, the NUEP of OF1 (4.2%) showed otherwise. The lack of synchronization between N released from organic sources and demand by the crop resulted in excess nutrient at the root zone that is vulnerable to leaching is the probable

cause (Evanylo *et al.*, 2008; Rosen & Allan, 2007). This was supported by Wyland *et al.* (1996) and Hartz (2006) which highlighted the nutrient loss from the organic system might be due to excessive nutrient enrichment from repeated applications of compost in organic systems.

Excessive nutrient can be washed off from the agricultural land by rainfall and irrigation and enters the water bodies (DEFRA, 2015a). The N losses per produce were highest in CF2 and OF1 with N loss fraction of 27 and 14 kg N per kg of produce, respectively. The N loss fraction in CF1 and OF2 were both 3 kg N lost per kg of produce. Several researches showed that only 20% to 68% of N applied in the field were taken by crop (Dungait *et al.*, 2012; Fortes *et al.*, 2011; Gardner & Drinkwater, 2009; Goulding *et al.*, 2008; Yan *et al.*, 2014). The higher N loss fraction in CF2 and OF1 indicated potential N pollution from the farms especially when excess N entered aquatic systems (Jarvis *et al.*, 2011; Rosen & Eliason, 1996). The nutrient in waterways can cause eutrophication, algal bloom while the pathogen attached on wash off manure or soil particles can easily contaminate the water bodies (Beddington *et al.*, 2012b).

#### **4.10.4 Waste generation rate**

Waste generation rate per kg of produce was about 0.01, 0.04, 0.07 and 0.04 kg kg<sup>-1</sup> in CF1, CF2, OF1 and OF2, respectively. Waste generation rate per unit of production in farms indicated highest waste produced in OF1 while the lowest was in CF1. The typical wastes found in the farms are crop residues and plastic packaging wastes and similar waste compositions were reported in NIEA (2012). Organic wastes were the major wastes generated in the farms in this study. Organic wastes emit methane and leachate when the waste starts to rot and decompose on the fields which contribute to GHG emissions (UNEP, 2009). The current waste management practices in all study

farms involved either storing or reusing the agriculture wastes within the farms premises.

There is no waste collection available in all the study farms. This is mainly due to low quantity of wastes generated from agriculture premises, the agriculture waste was often neglected. In addition, the study farms located around suburb and remote area which increased the difficulties for waste collection and transportation. Similar situations were reported by Hurley (2008) and Briassoulis et al. (2013) that the collections of agriculture wastes were difficult especially at remote area and it is more challenging when the transportation cost is high due to the long distance and bulk nature of the agriculture wastes. Other than that, limited recycling facilities and restriction on disposal of agriculture waste at municipal landfill were some of the factors for improper waste management in farms (Sonnevera, 2011). Thus, some farmers still prefer to accumulate or bury the agriculture wastes within farms premises.

#### **4.11 Cost-profit analysis**

Table 4.18 shows the cost profit analysis of each study farms. The average cost per unit of produce in CF1 is about RM 1.31 kg<sup>-1</sup> (≈USD 0.31 kg<sup>-1</sup>). The main cost in CF1 operation is the purchase of consumable product such as pesticide, herbicide, chicken manure, seeds and compost. The farm employed 10 workers and the salary costs around RM 144,000 (≈USD 34,212) per year. Lower average costs per unit was observed in CF2 which was about RM 1.23 kg<sup>-1</sup> (≈USD 0.29 kg<sup>-1</sup>) with the main expenditure from labour costs which were about RM 86,400 (≈USD 20,527) per year. Four workers were employed in CF2 and their salary is based on commission basis of RM 0.20 (≈USD 0.05) per kg produce. Consumable such as synthetic fertilizer, manure, agrochemical and seeds costs about RM 32,400 (≈USD 7,698) per year in CF2.

Table 4.18: Cost-profit analysis

<b>Input Section</b>	<b>CF1</b>	<b>CF2</b>	<b>OF1</b>	<b>OF2</b>
Expected Sales volume-units, kg y <sup>-1</sup>	610,876	108,000	12,610	203,908
Price per unit, RM kg <sup>-1</sup> y <sup>-1</sup>	1.50 (≈USD 0.36)	2.10 (≈USD 0.50)	13.00 (≈USD 3.10)	5.00 (≈USD 1.20)
Tourism income	Nil	Nil	NA	NA
<b>Fixed costs</b>				
Pesticide, herbicide, fertilizer, chicken manure, compost material, RM y <sup>-1</sup>	600,000 (≈USD 142,551)	32,400 (≈USD 7,698)	68,518 (≈USD 16,279)	125,479 (≈USD 29,812)
Diesel, RM y <sup>-1</sup>	9,163 (≈USD 2,177)	2,268 (≈USD 539)	1,639 (≈USD 389)	10,195 (≈USD 2,422)
Wages, RM y <sup>-1</sup>	144,000 (≈USD 34,212)	86,400 (≈USD 20,527)	54,000 (≈USD 12,830)	288,000 (≈USD 68,425)
Maintenance, RM y <sup>-1</sup>	45,816 (≈USD 10,885)	11,340 (≈USD 2,694)	8,196 (≈USD 1,947)	50,977 (≈USD 12,111)
<b>Contribution Margin</b>				
Revenue, RM y <sup>-1</sup>	916,313 (≈USD 217,703)	226,800 (≈USD 53,884)	163,925 (≈USD 38,946)	1,019,538 (≈USD 242,228)
Total costs, RM y <sup>-1</sup>	798,979 (≈USD 189,826)	132,408 (≈USD 31,458)	132,353 (≈USD 31,445)	474,651 (≈USD 112,771)
<b>Contribution Margin, RM y<sup>-1</sup></b>	<b>117,335</b> <b>(≈USD 27,877)</b>	<b>94,392</b> <b>(≈USD 22,426)</b>	<b>31,571</b> <b>(≈USD 7,501)</b>	<b>544,887</b> <b>(≈USD 129,458)</b>
Contrib. margin per unit, RM y <sup>-1</sup>	0.19 (≈USD 0.05)	0.87 (≈USD 0.21)	2.50 (≈USD 0.59)	2.67 (≈USD 0.63)
Contrib. margin ratio, %	13	42	19	53

Nil – No tourism activities involved

NA – Involved tourism activities but income information not available

The low yield in OF1 caused high production costs of RM 10.50 kg<sup>-1</sup> (≈USD 2.49 kg<sup>-1</sup>). The main cost was contributed by consumables (RM 68,518 ≈ USD 16,279) such as compost, pest repellent and peat moss and seeds. The employment of five workers costs RM 54,000 (≈USD 12,830). In comparison to OF1, the production cost was much lower in OF2 which is about RM 2.33 (≈USD 0.55) per kg of produce. The employment of 20 workers in OF2 contributed to major fixed costs of RM 288,000 (≈USD 68,425) while the consumable costs of compost, gypsum, seeds were about RM 125,479 (≈USD 29,812), annually.

In general, the production costs in organic systems were higher than conventional systems in this study. Similar results were reported whereby the organic systems production costs were 28-34% higher compared to the conventional system, eg. the fresh tomato costs USD 0.35 kg<sup>-1</sup> in conventional farms while in organic farm is about USD 0.54 kg<sup>-1</sup> (Brumfield *et al.*, 2000; Clark *et al.*, 1999; Uematsu & Mishra, 2012). Most of the costs in the organic farms were from labour cost which were about 41 and 61% of the total costs in OF1 and OF2. This was supported by Uematsu & Mishra (2012) who reported organic farms average spending on labour were \$310,000-\$361,000 greater than conventional farms.

The price of leafy vegetables in CF1 is RM 1.50 kg<sup>-1</sup> (≈USD 0.36 kg<sup>-1</sup>) which was the lowest among all farms. The market channel in CF1 is by selling to wholesaler (locally address as middle man) who collects vegetables from several farms and later trade at wholesale market. According to grower of CF1, the profit margin was low because of the instability of vegetable price offered by wholesaler and similar issues were also reported by Huong *et al.* (2013) and Shrestha *et al.* (2014). The yearly yield of 610,876 kg y<sup>-1</sup> had generated RM 117,335 (≈USD 27,877) profit margin. In contrast to CF1, the



growers of CF2 focused on selling their produce at a higher price of RM 2.10 kg<sup>-1</sup> (≈USD 0.50 kg<sup>-1</sup>) by targeting restaurant and also exporting to neighbouring country-Singapore. Even though the yield in CF2 is not as high as CF1, but the marketing strategies resulted with RM 94,392 (≈USD 22,426) contribution margin per annum.

In OF1 the average price for organic vegetable was RM 13 (≈USD 3.10) per kg. According to farmers in OF1, the home delivery service and direct selling to visitors who visited the farms allowed higher pricing on the produce. Even though the price per produce was highest in OF1 among the farms in this study, the total contribution margin estimated was RM 31,571 (≈USD 7,501) per annum which was the lowest among the farms. The farmers of OF1 explained the major issue in the farm was the low vegetable yield. Thus, main incomes in OF1 were from tourists visit but the profit contributions were not disclosed. In contrast to OF1, the produce price in OF2 was lower which is about RM 5.00 kg<sup>-1</sup> (≈USD 1.20 kg<sup>-1</sup>). The growers of OF2 opted variable marketing channels such as direct selling to consumer, retail shop, super and hypermarket, and export to Singapore. The marketing strategies in OF2 contributed total of RM 544,887 (≈USD 129,458) annual profit. The results revealed that the variation of yield, marketing channel and production costs leads to different profit margin between each study farm. The pricing of leafy vegetable varies between each farm due to different marketing channels used by the farmers. The marketing channel opted by organic farmers were more diversified and direct to consumer. This was supported by Kremen et al. (2004) which suggested organic producers tend to select market channels that are direct to consumer. The price of organic produce in farms in this study was 100 to 700% higher than conventional farms. This was supported by Brumfield et al. (2000) and Dimitri & Greene (2002) who reported that the organic produce price range from 100% and 250% of conventional vegetables. In some countries organic potatoes price can

range from 50% to more than 500 % higher as compared to conventional produce (Offermann & Nieberg, 2000). The premium price of organic produce was often seen as farming method that allowed higher income and profit for local farmers (Clark *et al.*, 1999; Scialabba & Müller-Lindenlauf, 2010; Shrestha *et al.*, 2014; Tudisca *et al.*, 2014). However, this is only true in OF2 which achieved highest contribution margin per unit among the farms in this study. The low yield in OF1 has resulted low profit even with higher produce pricing. Study by Rani *et al.* (2013) suggested that the incomes of organic farms were not significantly higher than conventional farms due to the low yield even with higher pricing for organic produce. Studies by Brumfield *et al.* (2000) and Uematsu & Mishra (2012) highlighted that the high production expenses, high labour requirement and lower marketable yields caused smaller revenue earned by the organic producers.

#### **4.12 General summary**

Table 4.19 summarized the agri-environment indicators that were assessed in this study. This study reveals not all organic farms are environmental friendly and not every conventional farm are unsustainable as the public perceived. In general, the agri-environment assessment of CF1 showed high farm efficiency and lesser environmental impact than CF2. The compliance of CF1 to good agricultural practice MyGAP certification might provide the information and tools needed to achieve sustainable farming. On the other hand, both organic farms were certified by local organic certification SOM. However, OF2 were also in compliance with international organic certification NASAA which has more stringent rule and regulation that might contribute to higher farm efficiency and lesser impact on environment in OF2 as compared to OF1. This suggests agriculture certification might play an important role in ensuring sustainable agriculture development.

Table 4.19: Agri-environment indicators

		<b>CF1</b>	<b>CF2</b>	<b>OF1</b>	<b>OF2</b>
Carbon sequestration potential	C balance (STAN model), kg C ha <sup>-1</sup> y <sup>-1</sup>	6,315 ± 2,530 (Sink)	6,315 ± 2,530 (Sink)	-304 ± 12,988 (Source)	10,811 ± 4,929 (Sink)
Nutrient management	N balance (STAN model), kg N ha <sup>-1</sup> y <sup>-1</sup>	1,589 ± 156 (Sink)	1,605 ± 8 (Sink)	2,608 ± 18 (Sink)	912 ± 220 (Sink)
Soil C and N stock	Soil carbon concentration, %	1.65 ± 0.289	2.63 ± 0.248	4.38 ± 0.673	2.62 ± 0.396
	Soil nitrogen concentration, %	0.209 ± 0.053	0.458 ± 0.106	0.355 ± 0.109	0.431 ± 0.263
Water management	Nitrate in runoff, mgL <sup>-1</sup>	20.0 ± 18.2	16.5 ± 10.7	26.7 ± 2.9	29.6 ± 23.6
	Nitrate in leachate, mgL <sup>-1</sup>	1.75 ± 0.55	50.3 ± 23.9	95.7 ± 17.1	86.7 ± 60.3
GHG emission	C emission, kg C ha <sup>-1</sup> y <sup>-1</sup>	1877	2156	18982	2541
	N emission, kg N ha <sup>-1</sup> y <sup>-1</sup>	139.1	24.6	142	33.9
Pathogen	Coliform/ <i>E.coli</i> contamination	+	+	+	+
	<i>Salmonella/Shigella</i> contamination	+	+	+	+
Farm efficiency	Yield, t ha <sup>-1</sup> y <sup>-1</sup>	610.8 ± 342	108 ± 12	12.6 ± 5.3	203.9 ± 156
	NUEP, %	17.7	2.3	4.2	27.6
	WUEP, kg m <sup>-3</sup>	1	0.03	2	7
	Waste generation rate per unit of produce, kg kg <sup>-1</sup>	0.01	0.04	0.07	0.04
Cost benefit analysis	Cost per kg produce, RM kg <sup>-1</sup>	RM 1.31 (≈USD 0.31)	RM 1.23 (≈USD 0.29)	RM 10.50 (≈USD 2.49)	RM 2.33 (≈USD 0.55)
	Contribution Margin, RM y <sup>-1</sup>	RM 117,335 (≈USD 27,877)	RM 94,392 (≈USD 22,426)	RM 31,571 (≈USD 7,501)	RM 544,887 (≈USD 129,458)

+: potential contamination

## CHAPTER 5: CONCLUSION

In this study, vegetable farms CF1 and CF2 are conventional while OF1 and OF2 are organic farms. The main characteristic that differentiates organic and conventional farms is used of chemical within farms systems. In organic farms, the used of synthetic chemical is forbidden whereas it is allowed to use in conventional farms according to country's rules and regulations. The C mass balance indicates that CF1, CF2 and OF2 were potential C sinks while OF1 was identified as C source. Based on STAN model, the negative C balance in OF1 was due to the high CO<sub>2</sub> emission. The results suggest high soil C input might favour microbial decomposition activities that lead to increased C emission from the farm system. The results highlight that the C emission may undermine the environmental benefit of C sequestration in organic system. The flow models suggest N surplus in all study farms, which may potentially accumulate in various soil fractions or become lost to the environment via volatilization, runoff, or leaching. Both runoff and leachate (except for leachate in CF1) in the study farms were above the USEPA limit for surface water for municipal use and drinking purpose, while the leachate samples from CF2, OF1 and OF2 had exceeded the EC drinking water standard limit. The results suggest that the high nitrate runoff and leachate which directly enters the surface and ground water might cause pollution.

GHG emissions were detected from all the soil in the study farms and CO<sub>2</sub> was identified as the major gas emitted. The GHG emissions rate vary from farm to farm. However, in general, organic farming systems emitted higher gaseous C than conventional systems. Study revealed that large volume of C input and high soil microbial count in OF1 suggested excessive use of compost might enhance soil respiration and microbial activity which increased C gaseous emission. The high gaseous emission in OF1 resulted in the highest GWP among the farms in this study

while the GWP of CF1, CF2 and OF2 were similar. This study suggests revision of nutrient regime in OF1 in order to reduce GHG emission and at the same time maintaining yield.

The water flow models highlighted that the recycling of water runoff through catchment pond can contribute to C sequestration and the use of water from catchment pond as irrigation can channel some of the nutrients back to the farm soil. Additionally, the catchment pond acted as riparian zone that retained, reduced or even removed the nutrient content in runoff. The recycling of runoff for irrigation is recommended to mitigate NPS pollution and increase WUEP.

Pathogens were found to be present in all the farm systems with different level of population in soil, vegetable and water runoff samples. However, no significant differences of pathogen level between organic and conventional farms were observed in this study. In the aspect of food safety, the pathogen count suggests potential contamination of vegetable product in the study farms and thus highlights the issue of food safety, especially for leafy green vegetables such as lettuce that is often consumed raw. Postharvest treatment such as washing is suggested to reduce pathogen level in vegetables.

The farm efficiency was evaluated based on yield, nitrogen use, water use and waste generation. Results indicate high yield in CF1 and OF2, however, the average yield in conventional systems was significantly higher than organic systems. The yield in OF2 doubled the yield in CF2 suggests that appropriate farm management allows organic systems to achieve comparable yield or even higher than conventional systems. The NUEP was ranked as OF2>CF1>OF1>CF2. In terms of water use, the highest

efficiency was in OF2 followed by OF1, CF1 and lastly CF2. The results of NUEP and WUEP indicate comparable farm efficiencies between CF1 and OF2. This further highlights farm management has larger influence over farm sustainability than types of farming systems.

The proportion of wastes output from CF1, CF2, OF1 and OF2 were generally low. This study reveals organic wastes and plastic wastes were the two major wastes category found in study that is available for reuse and recycle. The cost-profit analysis reveals that the general perception of higher income from organic producer is only applicable for OF2, thus suggests influence of individual farm management over the production revenue.

This study concludes farm level management plays an important role in achieving sustainable agriculture. Individual farm management varied even with similar farming systems. Thus, one organic farm can have better farm efficiency and lower environmental impact than the other organic farm and same goes for conventional farming. The results suggest both organic and conventional farming can be sustainable with appropriate management. Based on the study, it is suggested to recycle runoff and irrigation in the farms which increase water use efficiency and also recycled nutrient back to fields. In addition, farmers should reduce nutrient input in order to reduce nutrient loss from the farms. The study also demonstrates MFA and SFA with STAN software allow a comprehensive assessment of agri-indicators which offers an alternative method for farm evaluation. This provides information on the farm situation and potential imminent environmental concerns that is able to assist farmers or even policy makers in decision making process. Such assessment can also be the tool for Malaysian agriculture certification such as SOM or MyGAP. However, the MFA and

SFA models are limited to particular spatial and times, thus constant monitoring become paramount to evaluate the dynamic interrelation among soil fertility, biophysical characteristics, farm management, climate, and socio-economic factors.

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## LIST OF PUBLICATIONS AND PAPERS PRESENTED

### a) Publications

1) Hongyeng, L., & Agamuthu, P. (2014). Material/substance flow analysis of carbon flux in an organic and a conventional vegetable farm. *Pakistan journal of agricultural sciences*, 51(3), 511-516. (ISI Q2) (Appendices G)

2) HongYeng, L., & Agamuthu, P. (2015). Nitrogen flow in organic and conventional vegetable farming systems: a case study of Malaysia. *Nutrient Cycling in Agroecosystems*, 103(2), 131-151. (ISI Q2) (Appendices H)

### b) Paper presented in conference

1) "Material Flow Analysis of Organic Farm and Conventional Farm" International Conference on Waste Management 2013, 26th & 27th August 2013, Malaysia.

2) "Biomass Flow and Carbon Sequestration in an Organic Farm" 2014 ASEAN Conference on Science and Technology 18-19 August 2014, Bogor, Indonesia.

**Appendice A: C concentration of vegetables at farm**

Local name	CF1	CF2	OF1	OF2
Amaranth	34.82	33.90	33.54	33.54
Amaranth, Red	NA	NA	34.95	34.95
Choy Sum	37.67	31.77	NA	32.65
Fu Gui Choy	NA	NA	31.73	NA
Fu Mak	NA	NA	39.92	NA
Hong Kong Choy Sum	35.70	NA	30.91	NA
Japanese Choy Sum	NA	34.67	35.65	NA
Kai Lan	NA	NA	33.76	NA
Lettuce	NA	NA	35.93	35.93
Mini Cos Lettuce	NA	NA	34.55	NA
Nai Bai	NA	NA	34.05	34.05
New Zealand Spinach	NA	NA	38.67	38.67
Ong King Pak Choy	NA	NA	34.60	NA
Senposai	NA	NA	34.96	NA
Spring Onion	NA	36.88	NA	NA
Sweet Potato Leaf	NA	NA	41.05	39.68
Water Spinach	36.67	35.20	35.00	36.24
Xiao Pak Choy	NA	32.42	36.32	31.46
Xiu Zhen Choy Sum	NA	NA	33.18	NA

**Appendice B: N concentration of vegetables at farm**

	<b>CF1</b>	<b>CF2</b>	<b>OF1</b>	<b>OF2</b>
Amaranth	3.946	5.132	5.376	5.376
Amaranth, Red	NA	NA	3.993	3.993
Choy Sum	5.037	6.08	NA	5.104
Fu Gui Choy	NA	NA	5.523	NA
Fu Mak	NA	NA	2.7	NA
Hong Kong Choy Sum	5.526	NA	4.81	NA
Japanese Choy Sum	NA	4.869	5.779	NA
Kai Lan	NA	NA	5.72	NA
Lettuce	NA	NA	4.654	4.654
Mini Cos Lettuce	NA	NA	4.915	NA
Nai Bai	NA	NA	6.275	6.275
New Zealand Spinach	NA	NA	4.45	4.45
Ong King Pak Choy	NA	NA	6.206	NA
Senposai	NA	NA	4.689	NA
Spring Onion	NA	3.996	NA	NA
Sweet Potato Leaf	NA	NA	6.082	6.082
Water Spinach	4.429	4.18	4.943	3.063
Xiao Pak Choy	NA	5.888	6.332	5.495
Xiu Zhen Choy Sum	NA	NA	5.133	NA

**Appendice C: Rainfall physical chemical properties**

	<b>CF1</b>	<b>CF2</b>	<b>OF1</b>	<b>OF2</b>
<b>DO, mg/L</b>	7.812(0.959)	7.237(1.111)	7.163(0.186)	9.028(0.410)
<b>BOD5</b>	2.459(0.774)	2.437(1.421)	3.955(1.226)	2.653(1.254)
<b>pH</b>	6.022(2.425)	6.873(0.983)	5.287(1.389)	5.785(0.683)
<b>Temperature (° C)</b>	29.0(1.2)	31.73(0.838)	24.33(3.803)	34.90(4.207)
<b>TDS (ppm)</b>	33.7(32.4)	11.6(7.90)	4.59(0.515)	4.61(0.984)
<b>Conductivity (µs)</b>	67.53(64.16)	23.17(15.75)	35.81(31.11)	9.250(1.868)
<b>TSS (mg/L)</b>	0.020(0.021)	0.013(0.005)	0.010(0.008)	0.130(0.091)
<b>Turbidity (FAU)</b>	4.5(3.697)	4.0(0.816)	5.0(1.633)	36.50(25.38)
<b>COD (mg/L)</b>	9.00(5.657)	2.0(1.414)	57.0(32.57)	42.0(26.81)
<b>Chloride, (mg/L)</b>	2.104(1.201)	1.203(0.264)	1.237(0.344)	3.155(2.844)
<b>Hardness, Mg (mg/L)</b>	0.370(0.256)	1.197(0.957)	0.630(0.229)	0.930(0.681)
<b>Hardness, Ca (mg/L)</b>	1.160(0.113)	1.360(0.482)	1.220(0.340)	1.278(0.473)



**Appendice D: Irrigation physical chemical properties**

	CF1		CF2		OF1		OF2	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
DO, mg/L	5.146	1.542	4.250	1.543	6.880	0.440	6.348	1.225
BOD5	4.533	1.620	1.588	1.148	2.445	2.099	4.114	1.374
pH	7.033	0.212	6.665	0.781	6.943	0.042	6.786	0.577
Temperature (° C)	28.750	1.735	30.325	0.630	28.350	0.541	27.920	4.026
TDS (ppm)	165.750	43.458	73.150	47.693	15.800	0.667	70.420	26.617
Conductivity (µs)	333.500	87.911	146.400	94.756	31.775	1.497	141.820	54.312
TSS (mg/L)	0.030	0.009	0.060	0.016	0.035	0.018	0.133	0.083
Turbidity (FAU)	7.210	4.408	17.750	3.961	12.750	6.300	51.667	18.903
COD (mg/L)	21.167	19.416	11.750	7.496	13.750	11.300	17.000	7.842
Chloride, (mg/L)	12.378	3.913	10.953	3.365	2.728	0.867	7.825	2.012
Hardness, Mg (mg/L)	0.208	0.179	3.500	0.787	0.613	0.087	5.140	4.328
Hardness, Ca (mg/L)	0.027	0.030	16.400	3.779	1.448	0.285	18.700	6.647

**Appendice E: Runoff physical chemical properties**

	CF1		CF2		OF1		OF2	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
DO, mg/L	6.658	1.068	5.145	2.974	4.283	0.536	5.714	1.773
BOD5	3.519	0.853	3.923	3.060	3.500	0.181	3.450	0.767
pH	7.738	0.277	7.100	0.609	8.853	0.681	7.268	0.490
Temperature (° C)	28.788	2.781	30.488	0.508	30.767	0.125	28.720	4.798
TDS (ppm)	257.375	117.214	128.425	104.416	156.333	40.467	172.380	113.468
Conductivity (µs)	481.709	286.728	257.825	209.613	312.000	82.369	226.238	117.986
<b>TSS (mg/L)</b>	<b>0.157</b>	<b>0.119</b>	<b>0.263</b>	<b>0.081</b>	<b>0.250</b>	<b>0.107</b>	<b>0.390</b>	<b>0.173</b>
Turbidity (FAU)	35.429	24.724	79.250	19.588	67.667	29.511	115.200	54.062
COD (mg/L)	55.500	30.251	101.500	43.935	164.667	34.374	62.000	56.414
Chloride, (mg/L)	35.348	18.728	36.903	13.620	26.067	1.040	24.278	11.701
Hardness, Mg (mg/L)	21.700	7.337	15.325	9.826	12.033	5.816	23.340	10.896
Hardness, Ca (mg/L)	0.263	0.097	19.125	7.181	4.000	3.360	17.000	10.762

**Appendix F: Leachate physical chemical properties**

	CF1		CF2		OF1		OF2	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
DO, mg/L	5.967	1.050	6.840	0.161	5.203	0.141	6.827	0.075
BOD5	3.077	1.678	6.327	0.341	4.133	0.182	3.280	0.596
pH	7.327	0.473	6.523	0.550	7.350	0.104	7.393	0.111
Temperature (° C)	28.667	1.041	31.467	0.680	30.567	0.713	26.167	5.173
TDS (ppm)	378.000	298.568	864.667	237.237	1016.000	199.304	991.333	337.758
Conductivity (µs)	756.667	593.689	1726.333	470.390	4.037	0.673	1600.000	70.711
TSS (mg/L)	0.065	0.035	0.347	0.056	0.235	0.125	0.993	0.921
Turbidity (FAU)	17.000	5.657	66.667	40.803	41.333	10.625	137.500	48.790
COD (mg/L)	41.333	24.583	106.000	12.728	173.667	61.299	215.667	60.119
Chloride, (mg/L)	21.667	3.657	69.700	10.801	276.333	15.585	62.867	18.932
Hardness, Mg (mg/L)	14.380	12.164	21.533	4.445	246.067	21.369	18.233	10.499
Hardness, Ca (mg/L)	1.033	0.907	0.953	0.132	1.033	0.759	1.100	0.964

## Appendix G: Publication 1

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### MATERIAL/SUBSTANCE FLOW ANALYSIS OF CARBON FLUX IN AN ORGANIC AND A CONVENTIONAL VEGETABLE FARM

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Organic input in agricultural farm system is thought to enhance carbon sequestration by increasing soil organic matter content. Yet, the findings on the soil carbon concentrations increment in organically managed soil still remain controversial. In this paper, carbon flow of the key processes in organic farm (OF) and conventional farm (CF) were modelled using material flow analysis (MFA) for evidence of carbon stock. Carbon flux modelling shows  $363 \text{ tC h}^{-1} \text{ y}^{-1}$  of entry into farm system and signify the potential of OF as carbon sink. On the other hand, CF was identified as carbon source because the farm had  $-34,724 \text{ tC h}^{-1} \text{ y}^{-1}$  carbon stock change. The carbon flux of OF system led to 29% increase in the soil carbon concentration, whereas at CF the carbon concentration decreased by 11.7%. High amount of carbon exit in OF and CF through surface runoff and leaching, hence, improvement of farm management is needed especially in water management. Lower water outflow was observed at OF than within CF; however, the high carbon concentration found in surface runoff and leaching indicates that carbon get washed off from the farm. Based on MFA results, the farm management for OF and CF system can be improved to ensure economic and environmental benefits.

**Keywords:** Greenhouse gases, carbon flux, organic farming, vegetable culture, carbon sequestration

#### INTRODUCTION

One-third of the global greenhouse gas emissions (GHG) come from agriculture sector (Gilbert, 2012). Proper land use management mitigate GHG emission and or even create carbon sink by encouraging carbon sequester practices (Vleeshouwers and Verhagen, 2002; Freibauer *et al.*, 2004; Ogle *et al.*, 2005; Luo *et al.*, 2010). IPCC has identified biomass application as the promising tool to capture and store carbon at terrestrial reservoir (Sims, 2007). Farming practices affect farm input which is the key factor for soil organic matter turnover rates that exert high influence over soil carbon content (Freibauer *et al.*, 2004). Organic farm (OF) is believed to be carbon (C) sequester because organic fertilizer application is the common practice in OF. While conventional farm's (CF) lack of organic input is viewed to be the contributor to GHG emission. Various reports tried to conclude the benefit of converting from CF to OF in regards to C sequestration. Evidence of higher soil C concentration in organically managed farm was found, yet some other studies have not agreed with such findings (Janzen, 2006; Leifeld and Fuhrer, 2010; Scialabba and Müller-Lindenlauf, 2010). Several modelling studies reveal that conversion of CF to OF increases soil C is only a temporary solution for C sequestration due to high GHG emission (Foereid and Høgh-Jensen, 2004). Carbon sequestration at tropics and subtropics region faced difficulties because of the high soil degradation rate (Lal, 2004). Thus, the restoration of degraded soil and ecosystems in tropics and subtropics is much needed. Extensive researches of carbon sequestration

have been done but few at tropical region (Foereid and Høgh-Jensen, 2004; Ogle *et al.*, 2005). The insufficient information of carbon storage on agricultural land is prevalent within developing world, tropics and subtropics region (Govaerts *et al.*, 2009). Data limitation is the main set back in meta-analysis of global soil carbon change (Leifeld and Fuhrer, 2010). This study discussed and provided an insight to carbon flux at tropical region and the variation from other studies.

Material flow analysis (MFA) is an integrated modelling and assessment tool to evaluate the environment sustainability, especially, waste management. However, it is not used in agri-environmental assessment. Therefore, this study utilized MFA for carbon modelling within the farm systems in order to have a comprehensive assessment of farm system sustainability. It is a practical analytical method to quantify flows and stocks of materials or substances in a defined spatial and system which provide vital information on farm system stability (Baccini and Brunner, 2012; Brunner and Rechberger, 2004). In addition, MFA highlights the existing and potential material stocks accumulating within a system which can cause environmental problems or a potential source of resources. Material flow emphasizes on the imminent resource and environmental issue without depending on indicators of environmental. This study evaluated the role of material flow modelling in understanding carbon flux dynamics of OF and CF system. The objectives were to identify the system differences between OF and CF, potential drivers for changes and differences between systems in carbon flow. It also aimed to

## Nitrogen flow in organic and conventional vegetable farming systems: a case study of Malaysia

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**Abstract** High levels of nitrogen (N) are typically used in leafy vegetable farms to maximize production. However, such practice often leads to nutrient pollution. Hence, N balance in intensive leafy vegetable farm production must be explored to improve current farm management practices and to avoid environmental pollution. This study aimed to generate partial N balance in two organic (OF1 and OF2) and two conventional (CF1 and CF2) vegetable farms by employing material flow analysis/substance flow analysis in the STAN modeling software. Results showed that 31,556, 32,798, 19,498, and 19,337 t ha<sup>-1</sup> y<sup>-1</sup> of materials entered CF1, CF2, OF1, and OF2, respectively, and contributed to the nitrogen surplus levels of 1577, 1667, 2953, and 961 kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively. The STAN model revealed the presence of N surplus in the organic and conventional systems used in the study.

**Keywords** Leafy vegetables · Nutrient balance · Nitrogen · STAN software · Organic and conventional farms

### Introduction

Agricultural activities are important sources of nitrogen (N) pollution, which affects surface water and groundwater and consequently leads to serious economic, environmental, and human health issues. About 140 million tons of N is lost to the environment globally (Qiu 2013). By 2050, world fertilizer consumption would increase further by 50 % while global N losses to the environment would increase by 70 % (Sutton and Bleeker 2013). In addition to obvious environmental concerns, the discharge of N through surface runoff and leaching represents an economic loss of nutrients, which tends to increase farm production cost. Overcoming environmental concerns while improving farm production efficiency is thus a major challenge.

Researchers have emphasized the importance of improving nutrient management, reducing nutrient losses, and recycling nutrients (Zhao et al. 2010; Sutton and Bleeker 2013). A detailed and quantitative understanding of nutrient balance in various farming systems is a prerequisite for achieving proper nutrient management. However, tracing the path of N through environmental reservoirs is a considerable challenge because of the complex N cycle, complicated oxidation stages and mechanism conversion, variety of interspecies, and intricate transport/storage processes (Galloway et al. 2004).

Soil nutrient balance is the difference between nutrient inputs (fertilizer, manure, and compost) and

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