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Agri-environmental Policies to Meet Consumer Preferences in Japan: An Economic-Biophysical Model Approach

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ABSTRACT

Promoting environmentally friendly farming products is crucial to meeting consumer demand. Although governments implement policy measures to improve the environmental performance of the agriculture sector, their impacts are difficult to assess. This study analyses the performance of agri-environmental policies in Japan, by using the OECD's policy impact model and reference level framework. In particular, it identifies the environmental impacts of three simulated agri-environmental policies based on farms' characteristics. The results suggest that a policy mix of regulation and an incentive payment would reduce environmental impacts, suggesting that targeted approaches could improve the cost-effectiveness of agri-environmental policies.

Keywords: Agri-environmental policies; Water quality; Climate change; Economic-biophysical model; Reference levels

1 Introduction

Environmental issues are rising in importance when consumers make decision on purchasing food (Grunert, 2011; Banterle *et al*., 2012; Banterle and Ricci, 2013). A 2007 survey, for example, showed that more than 90% of Japanese consumers prefer to purchase agricultural products produced in an environmentally friendly way* under certain conditions† (MAFF, 2007). According to this survey, approximately 30% of respondents purchase eco-friendly products since they want to contribute to environmental conservation. The Japanese government's public opinion poll in 2008 also revealed an increasing expectation about the environmental role of agriculture in the future. Almost half (48.9%) of respondents expected agriculture and rural areas to preserve biodiversity and provide landscapes, while one in three (29.6%) expected water resources to be preserved and natural disasters such as landslides and floods to be prevented (CAO, 2008).

These results clearly indicate that promoting environmentally friendly farming is crucial to meeting consumer demand as well as achieving a sustainable food system. However, although governments implement various agri-environmental policy measures in order to improve the environmental performance of the agriculture sector, their impacts on the environment are difficult to assess because of the existence of complex biophysical processes and heterogeneous agricultural and environmental conditions.

OECD (2010) and Sasaki (2012) analysed the linkages between agricultural policies and environmental

* For example, organic products or agricultural products produced with a 50% reduction in the amount or usage of chemical fertiliser or synthetic agricultural chemicals (MAFF, 2007).
† Such as the existence of neighbouring markets, reliable labelling systems, and cheaper prices (MAFF, 2007).

effects in Japan. In these studies, a decision-making economic model for representative farms was combined with a stylised site-specific biophysical model, which quantifies how different policy measures influence agricultural production practices and environmental effects (e.g. nitrogen (N) runoff and greenhouse gas (GHG) emissions). This OECD Stylised Agri-environmental Policy Impact Model (SAPIM) estimated government budget outlays and social welfare, based on a monetary valuation of the environmental effects, as well as crop production.

However, their models assumed a single farm type (business farm household) that operates six hectares of farmland, which is relatively large compared with the average farm size (ca. 2.1 ha) (MAFFSTAT). In addition, Japan is a mountainous island archipelago formed through volcanic action. Two-thirds (66%) of its area is forest, and only 30% of land is suitable for agriculture or urban use (OECD, 2009). Incorporating these hilly and mountainous Japanese lands into a model is thus important in order to understand the environmental impacts associated with agriculture and to develop suitable agri-environmental policies.

Moreover, agri-environmental policies should be compared under a policy framework. For instance, OECD (2001) established a reference level framework that compared the degree to which agri-environmental policies were achieving their environmental targets and the distribution of costs for achieving these targets among stakeholders. Because markets for agri-environmental services in which consumers can express their demand are non-existent (Cooper et al., 2009), discussion on sharing the costs of managing agri-environmental issues is essential. However, despite its usefulness, policy simulations with modelling approaches based on this framework have been limited.

Therefore, by building on this stream of previous research, this study analyses the degree to which agricultural policies meet consumer demand for environmentally friendly products in Japan, particularly by incorporating the following points:

- Differentiating farms' characteristics: The present study divides farm types into five categories: business farm household in non-mountainous areas (BFH-N-M) (average size: 6.1 ha), business farm household in mountainous areas (BFH-M) (4.3 ha), non-business farm household in non-mountainous areas (N-BFH-N-M) (1.3 ha), non-business farm household in mountainous areas (N-BFH-M) (0.9 ha), and the average farming household (AVERAGE) (2.1 ha). The related farming costs and other parameters are also categorised into these five groups.
- Including different environmental impacts depending on farm type: Different runoff rates are used for farms in non-mountainous and mountainous areas since N runoff rates differ depending on the slope of the farmland (see Yamamoto et al., 2008).
- Addressing new agri-environmental policy scenarios that have not been covered in previous works: Three policy scenarios (regulation, environmental payment, and the combination of both) are selected in this study in order to examine who should be remunerated and who is liable for charges, based on the OECD reference level framework (OECD, 2001).

Overall, this study aims to identify suitable agri-environmental policies for improving environmental effects (e.g. water quality and GHG emissions), taking into consideration different farm characteristics (e.g. farm size, slope of farmland). The structure of the remainder of this paper is as follows. Section 2 presents the theoretical framework. Section 3 explains the application of the empirical framework. Section 4 discusses the reference level framework used in the policy simulations. Section 5 provides the results of the policy simulations, and Section 6 concludes the discussion.

2 Theoretical framework

In the SAPIM, land is divided into rectangular parcels of the same size (0.1 hectare) and of homogeneous land quality (productivity: *q*; although land quality differs by parcel). Land use is classified into rice, upland crop production (or cultivation), and abandonment in this model; wheat is assumed to be an upland crop. In the model, the optimal land use allocation and N application (the combination of chemical and organic fertilisers) are considered based on generating private profits to farmers and social profits by taking environmental externalities into consideration.

The model consists of the following functions: 1) profit functions with quadratic nitrogen response functions for cultivated crops (rice and wheat), 2) exponential N runoff and purification functions, 3) GHG emissions functions that combine the CH_4, N_2O, and carbon sequestration functions and 4) social welfare functions. Farmers' profits are maximised under exogenous crop prices and input costs. The model thus estimates social welfare, crop production, N runoff, and GHG emissions under different policy scenarios.

2.1 Land use allocation

This study follows the approaches taken by OECD (2010) and Sasaki (2012). Let *G(q)* denote the cumulative distribution of *q* ($0 \leq q \leq 1$), while *g(q)* is its density. We assume that *g(q)* is continuous and differentiable. The total amount of land in the region is

$$G = \int_0^1 g(q)dq \tag{1}$$

It is assumed that only rice and wheat are cultivated in this region, *i* = 1,2. Let $L_j(q)$ denote the share of land of quality *q* allocated to crop *i*. Then, the total amount of land allocated to rice and wheat is given by

$$H_i = \int_0^1 L_i(q)g(q)dq \text{ , i} = 1{,}2 \tag{2}$$

Land abandonment is not considered in the theoretical part in order to simplify the discussion.

2.2 Profit function

It is assumed that rice and wheat are produced under constant returns to scale. The output of each crop per unit of land area is denoted by y_i, while yield is a function of land quality *q* and the application of N fertiliser to the agricultural field x_i (amount of application depends on land quality *q*), $y_i = f^i(x_i(q), q)$. The applied amount of fertiliser x_i is the combination of chemical fertiliser x_{ci} and organic fertiliserx_{oi}. This production function is increasing and concave in fertiliser and land quality, that is $f^i_{x_i} > 0, f^i_{x_ix_i} < 0,$ and $f^i_q > 0, f^i_{qq} < 0$. Further, we assume that arable land can be allocated to either rice or wheat. The profit from agricultural production is thus expressed as

$$\pi^i = p_i f^i(x_i(q), q) - cx_i(q) \;\; \text{for } i = 1{,}2 \tag{3}$$

Here, p_i refers to the price of crops and *c* to the N fertiliser price, which are both taken as given.

The use of organic fertiliser increases yields depending on the application: $\Phi^i(x_{oi}(q))$, defined as 1< $\Phi^i(x_{oi}(q))$ with $\Phi^i_{x_{oi}} > 0$ and $\Phi^i_{x_{oi}x_{oi}} < 0$. At the same time, the additional costs of collecting, transporting, and spreading organic fertiliser are incorporated into the profit function. In the presence of this yield-increasing effect and the additional costs of organic application, profit function (3) is modified as follows:

$$\pi^i = p_i f^i(x_{ci}(q), x_{oi}(q), q)\Phi^i(x_{oi}(q)) - c^i(x_{ci}(q), x_{oi}(q)) \;\; \text{for } i = 1{,}2 \tag{4}$$

2.3 Nitrogen runoff and purification function

Two environmental effects are assumed: water quality impacts through chemical fertiliser runoff and GHG emissions through the application of chemical and organic fertilisers. Aggregate N runoff is a function of chemical fertiliser use. Suppose that the N content in organic fertiliser is not included in the N runoff function, because this N content is only a serious problem for very large applications. The runoff of nutrients (kg) from each parcel is expressed as a function of the amount of chemical fertiliser applied x_{ci}:

$$z_i = v_i[x_{ci}(q)] \text{ for } i = 1{,}2 \tag{5}$$

with $v_x > 0$ and $v_{xx} > 0$. Thus, the runoff function is convex in fertiliser application. It is well known that paddy fields improve local water quality by removing nitrogen through denitrification and absorption. When the total nitrogen inflow in the paddy field water exceeds the total outflow of nitrogen discharged from the paddy field water, the paddy field works as a nitrogen removal site, which means z_i is negative. Then, the total runoff from the land area devoted to rice and wheat is

$$z = \int_0^1 \{v_1[x_{c1}(q)]L_1(q) + v_2[x_{c2}(q)]L_2(q)\}g(q)dq \tag{6}$$

The monetary valuation of runoff damage (purification benefit) is defined by a valuation function, *D(z)*, which is assumed to be convex ($D(z)_x > 0, D(z)_{xx} > 0$).

2.4 GHG emissions and sequestration function

Regarding GHG emissions, agriculture is an important anthropogenic source of CH_4 and N_2O. In addition to GHG emissions, agricultural soils serve as a carbon sink.

2.4.1 CH_4 emissions

The impact of organic fertiliser for CH_4 emissions is critical (Yan et al., 2005), and the amount of the applied material and CH_4 emissions can be described by a response curve. CH_4 is generated if soil is maintained in an anaerobic state. Because upland soils are normally oxidative and not in an anaerobic state, CH_4 is not produced. CH_4 emissions are thus denoted as

$$CH_4 = \int_0^1 m[x_{o1}(q)]L_1(q)g(q)dq \tag{7}$$

with $m_x > 0$ and $m_{xx} < 0$. Thus, the runoff function is assumed to be concave in the application of organic fertiliser (Yan et al., 2005; IPCC, 2006).

2.4.2 N_2O emissions

Following the guidelines of the Intergovernmental Panel on Climate Change, N_2O emissions are assumed to be a combination of direct emissions (denitrification) and indirect emissions (associated with atmospheric deposition and N runoff):

$$N_2O = \int_0^1 \{n_1[x_{c1}(q), x_{o1}(q), z_1]L_1(q) + n_2[x_{c2}(q), x_{o2}(q), z_2]L_2(q)\}g(q)dq \tag{8}$$

2.4.3 Carbon sequestration

Soil carbon stock is affected heavily by fertiliser management. The appropriate amounts of organic fertiliser can increase the soil carbon content and reduce total GHG emissions. The carbon sequestration function is given by

$$Seq = \int_0^1 \{s_1[x_{o1}(q)]L_1(q) + s_2[x_{o2}(q)]L_2(q)\}g(q)dq \tag{9}$$

with $s_x > 0$ and $s_{xx} < 0$. Thus, the sequestration function is concave in the application of organic fertiliser. Consequently, net GHG emissions are expressed as follows:

$$\begin{aligned} e = &\int_0^1 m[x_{o1}(q)]L_1(q)g(q)dq \\ &+ \int_0^1 \{n_1[x_{c1}(q), x_{o1}(q), z_1]L_1(q) + n_2[x_{c2}(q), x_{o2}(q), z_2]L_2(q)\}g(q)dq \\ &- \int_0^1 \{s_1[x_{o1}(q)]L_1(q) + s_2[x_{o2}(q)]L_2(q)\}g(q)dq \end{aligned} \tag{10}$$

The monetary valuation of emissions damage (sequestration benefit) is defined by the valuation function, $GW(e)$, which is assumed to be concave ($GW(e)_x > 0, GW(e)_{xx} < 0$).

2.5 Social welfare function

Chemical fertiliser affects both yield and environmental externalities. Moreover, there is a trade-off in the application of organic fertilisers: while organic fertiliser can help maintain soil fertility (yield-increasing effect) and increase carbon sequestration, it can increase CH_4 emissions and be a source of water quality problems. Therefore, a socially appropriate amount of N application must be decided by considering these effects. The social welfare maximisation problem can now be expressed as

$$SW = \int_0^1 \sum_{i=1}^2 \left[p_i f^i(x_i(q), q) \Phi^i(x_{oi}(q)) \quad c^i x_i(q)\right] L_i(q) g(q) dq \quad D(z) \quad GW[e(m, n, s)]$$
(11)

Social planners choose the use of inputs (chemical and organic fertilisers) for each parcel under heterogeneous land productivity levels. The first-best optimum is solved as follows:

$$SW^i_{x_{ci}} = p_i f^i_{x_{ci}} - c^i_{x_{ci}} - D'(z)\frac{\partial v_i}{\partial x_{ci}} - GW'(e)\frac{\partial n_i}{\partial x_{ci}} = 0 \tag{12}$$

$$SW^i_{x_{oi}} = p_i f^i_{x_{oi}} \Phi^i_{x_{oi}} - c^i_{x_{oi}} - D'(z)\frac{\partial v_i}{\partial x_{oi}} - GW'(e)\left[\frac{\partial m_i}{\partial x_{oi}} + \frac{\partial n_i}{\partial x_{oi}} - \frac{\partial s_i}{\partial x_{oi}}\right] = 0 \tag{13}$$

Based on the optimal use of inputs and thus profits for each crop from a given land quality, land is allocated to the highest social return use in each parcel. The unique value of switching land quality q_1 is defined as

$$\pi_1^*(q_1) - D_1^*(q_1) - GW_1^*(q_1) = \pi_2^*(q_1) - D_2^*(q_1) - GW_2^*(q_1) \tag{14}$$

Consequently, land is allocated to crops by taking account of not only profits, but also the effect of the land allocation on N runoff and GHG emissions. The private optimum can easily be extracted from equations (11) to (14), under which the farmer ignores the effects of environmental externalities. By setting the marginal damage (benefit) to zero, $\pi_1^*(q_1) = \pi_2^*(q_1)$ is obtained.

3 Empirical framework

As explained in the Introduction, this study categorises farms into five types: business farm household in non-mountainous areas (j=1), business farm household in mountainous areas (j=2), non-business farm household in non-mountainous areas (j=3), non-business farm household in mountainous areas (j=4), and average farm household (j=5), based on farm size data provided by MAFFSTAT. We follow OECD (2010) and Sasaki (2012) to estimate the profit function, nitrogen response function, N runoff and purification function, GHG emissions and sequestration function, and social welfare function for each farm type.

3.1 Profit function

Farmers' profits from production in the absence of government intervention are given by

$$\pi^{ij} = p_i y_{ij} - c x_{ij} - o_{ij} \quad \text{for } i = 1,2 \text{ and } j = 1,2,3,4,5 \tag{15}$$

where p_i refers to the price of crops, y_{ij} to the yield/0.1 ha, c to the fertiliser (nitrogen) price, x_{ij} to the amount of N application, and o_{ij} to other costs.

The model employs a quadratic nitrogen response function, $y_{ij} = a_{ij} + \alpha_{ij} x_{ij} - \beta_{ij} x_{ij}^2$, estimated for crop 1 (rice) and crop 2 (wheat) for each farm type (j=1,2,3,4,5). When farmers consider using organic fertiliser x_{oij} in addition to (or instead of) chemical fertiliser x_{cij}, total N application x_{ij} is equal to the summation of the N content of the fertiliser and organic matter. The N content in chemical fertiliser is assumed to be 21–46% based on MAFFSTAT. Similarly, Okayama Prefecture (2008) set the average N content in organic fertiliser as 0.7% and the yield efficiency of organic fertiliser as 60% compared with 100% for chemical fertiliser.

It is assumed that the positive impact of using 1 tonne of organic matter on yields is $\Phi^i(x_{oij})$, while that of paddy is assumed to be 5% and that of wheat 10%, based on field survey data (e.g. NARO, 2007; Shibahara et al., 1999). Taking into consideration the additional cost for the application of organic matter, profit function (15) is now expressed as follows:

$$\pi^{ij} = p_i(a_{ij} + \alpha_{ij}x_{ij} - \beta_{ij}x_{ij}^2)\Phi'(x_{otj}) - cx_{otj} - (c_{op} + c_{ot} + c_{os})x_{otj} - o_{ij}$$

$$\text{for } i = 1,2 \text{ and } j = 1,2,3,4,5 \tag{16}$$

where c_{op} refers to the price of organic matter (JPY/tonne), c_{ot} to transportation cost (JPY/tonne), and c_{os} to spreading cost (JPY/tonne). The parameters for the model are reported in Table 1.

Table 1.
Parameter values in the numerical application

Parameter	Symbol	Value	Parameter	Symbol	Value
Price of crop:		JPY/kg	**Other cost:**		JPY/0.1ha
Rice	P_1	194.3	**Rice**		
Wheat	P_2	50.6	BFH-N-M	O_{11}	61 891
Price of chemical fertiliser (nitrogen)		JPY/kg	BFH-M	O_{12}	71 614
	C	363	N-BFH-N-M	O_{13}	77 721
Organic matter:		JPY/t	N-BFH-M	O_{14}	115 813
Price of organic matter	C_{op}	5 000	AVERAGE	O_{15}	70 570
Transportation cost	C_{ot}	1 000	**Wheat**		
Spreading cost	C_{os}	2 000	BFH-N-M	O_{21}	37 113
			BFH-M	O_{22}	42 818
			N-BFH-N-M	O_{23}	39 107
			N-BFH-M	O_{24}	42 014
			AVERAGE	O_{25}	38 073

Source: MAFFSTAT, MAFF(2008)

3.2 Nitrogen response function

3.2.1 *Rice paddy*

The quadratic nitrogen response function of rice paddy was estimated by OECD (2010) and Sasaki (2012), based on Toriyama (2002) as follows:

$$y_1 = 368.6 + 31.7x_1 - 1.4x_1^2 \tag{17}$$

Land quality *q* and farm types *j* are incorporated into response function (17) through the parameters α_{1j} and β_{1j} by calibrating the response function to reflect the yield spread in the field survey data of Toriyama (2002). Thus, the quadratic nitrogen response function of rice paddy is expressed as

$$y_{1j} = 368.6 + \alpha_{1j}x_{1j} - \beta_{1j}x_{1j}^2$$

$$\text{where } 22.19 \leq \alpha_1 \leq 41.21,\ 0.98 \leq \beta_1 \leq 1.82, \tag{18}$$

$$\alpha_{1j} = e_{0j} + e_{1j}q, \quad \beta_{1j} = \mu_{0j} + \mu_{1j}q$$

When *q* is distributed uniformly among parcels, parameters $e_{0j}, e_{1j}, \mu_{0j},$ and μ_{1j} are estimated (Table 2).

Table 2.
Rice paddy parameter values in different farm types

	e_{0j}	e_{1j}	μ_{0j}	μ_{1j}
BFH-N-M	21.87	0.32	0.97	0.01
BFH-M	21.74	0.45	0.96	0.02
N-BFH-N-M	20.61	1.56	0.91	0.07
N-BFH-M	19.81	2.38	0.88	0.11
AVERAGE	21.24	0.95	0.94	0.04

Source: Authors' calculation.

3.2.2 Wheat

The quadratic nitrogen response function of wheat (converted from rice cultivation) was estimated by OECD (2010) and Sasaki (2012) by using the datasets of the National Agricultural Centre as follows:

$$y_2 = 214.9 + 45.6x_2 - 1.2x_2^2 \tag{19}$$

Data that can estimate the quadratic nitrogen response function of wheat is limited in Japan. National Agricultural Centre data are based on the highest yields from research fields (maximum yield 700–800 kg/0.1 ha). Therefore, function (19) may not be a representative average response function and thus wheat yield is adjusted based on average yield data depending on farm size (the national average wheat yield is 388 kg/0.1 ha; MAFFSTAT). For this, we use the multiplier Λ_j and calibrate the response function to reflect the yield spread. Land quality *q* and farm types *j* are incorporated into response function (19) through the parameters α_{2j} and β_{2j}, making the quadratic nitrogen response function of wheat

$$y_{2j} = 214.9 + \alpha_{2j}x_{2j} - \beta_{2j}x_{2j}^2$$

$$\text{where} \quad 11.4\Lambda_j \le \alpha_{2j} \le 45.6\Lambda_j,\ 0.3\Lambda_j \le \beta_{2j} \le 1.2\Lambda_j, \tag{20}$$

$$\alpha_{2j} = h_{0j} + h_{1j}q\,,\ \beta_{2j} = \eta_{0j} + \eta_{1j}q$$

Subsequently, each parameter is obtained as shown in Table 3.

Table 3.
Wheat parameter values in different farm types

	Λ_j	h_0	h_1	η_0	η_1
BFH-N-M	0.9	9.75	0.51	0.26	0.01
BFH-M	0.8	8.47	0.65	0.22	0.02
N-BFH-N-M	0.5	4.28	1.43	0.11	0.04
N-BFH-M	0.4	2.85	1.71	0.08	0.05
AVERAGE	0.6	5.81	1.03	0.15	0.03

Source: Authors' calculation.

3.3 Nitrogen runoff and purification function

3.3.1 Rice paddy

It is difficult to describe the relationship between N application and its environmental impact because N runoff from irrigation and meteoric water might affect the nitrogen balance in paddy fields. Taking into consideration the large effect of fertiliser application on N runoff (Kunimatsu and Muraoka, 1989), the relationship between N application and runoff is estimated by using field survey data collected by Shiga Prefecture in 2007:

$$z_{1j} = 0.0062e^{0.1468x_{1j}} - 1.14 \tag{21}$$

where z_{1j} refers to N runoff or purification and x_{1j} to the application amount of N. Paddy fields either serve as N removal or pollution sites depending on the agricultural activities and nitrogen concentration of irrigation water.

3.3.2 Wheat

Because soil condition, crops, cropping season, and methodological conditions all affect N runoff, the exponential relationship between N application and N runoff is estimated based on Japanese field data collated by the National Institute for Ago-Environmental Science:

$$z_{2j} = 1.129e^{0.114x_{2j}} \tag{22}$$

where z_{2j} refers to N runoff and x_{2j} the application amount of N.

3.3.3 Nitrogen runoff and slope

To reflect the N runoff–farmland slope relationship, this study introduces the parameter φ_s. Thus, functions (21) and (22) can be rewritten as

$$z_{1j} = \varphi_s(0.0062e^{0.468x_{1j}} - 1.14) \tag{23}$$

$$z_{2j} = \varphi_s(1.129e^{0.114x_{2j}}) \tag{24}$$

Yamamoto et al. (2008) studies the relationship between slope and N runoff in mountainous areas and estimates the linear relationship as

$$\mathrm{R} = 0.0059\mathrm{d} + 0.0612 \tag{25}$$

where R refers to N runoff and d to slope degree. In Japan, the slope of farmland (both rice paddy and upland) is classified into two categories: steep (2.86 degrees for rice paddy and 15 degrees for upland) and gentle (0.57 degrees for rice paddy and 8 degrees for upland). Subsequently, φ_s is obtained as presented in Table 4, and multiplied for business farm household in mountainous areas and non-business farm household in mountainous areas.

Table 4.
Slope parameter values

	φ_s
Steep slope rice paddy	1.28
Gentle slope rice paddy	1.06
Steep slope upland	2.45
Gentle slope upland	1.77

Source: Authors' calculation.

3.4 GHG emissions and sequestration function

Rice cultivation is the main anthropogenic source of CH_4 emissions. Fertiliser application and ploughing organic soil create ammonium ions inside the soil, leading to N_2O emissions. In addition to CH_4 and N_2O emissions, agricultural soils serve as a carbon sink. Net GHG emissions are therefore explained by converting CH_4 and N_2O into CO_2 equivalent (tonne) (IPCC, 2006):

$$\mathrm{GHG}(CO_2eq) = 21CH_4 + 310N_2O + CO_2 \tag{26}$$

3.4.1 CH_4 emissions

OECD (2010) and Sasaki (2012) estimated CH_4 emissions (t CH_4/0.1 ha/yr) as follows:

$$CH_4 = 0.001296 \times (1 + 1.4x_{ot})^{0.59} \tag{27}$$

In this study, the estimated values of CH_4 and x_{Ut} depend on farm type *j*. However, in upland areas, CH_4 is not generated because such soils are normally oxidative and aerobic.

3.4.2 N_2O emissions

OECD (2010) and Sasaki (2012) estimated the N_2O emissions associated with fertiliser application as

$$N_2O_{direct\_i} = \frac{1}{1000} \times EF_{di} \times x_i \times \frac{44}{28} \tag{28}$$

where $N_2O_{direct\_i}$ refers to direct N_2O emissions derived from fertiliser application in land use i (t N_2O), 1/1000 is the conversion of units from kg to tonne, EF_{di} is the emissions factors (kgN_2O-N/kgN) (for paddy: 0.0031; for upland crops: 0.0062), and x_i is the amount of N application (kgN). Moreover, 44/28 represents the conversion of N_2O-N emissions into N_2O emissions. In this study, the estimated values of $N_2O_{direct\_i}$ and x_i depend on farm type j.

Further, when E_{adi} is N_2O emissions associated with atmospheric deposition (kgN_2O) and E_{li} emissions associated with N leaching and runoff (kgN_2O), indirect emissions $N_2O_{indirect\_i}$ is expressed as follows:

$$N_2O_{indirect\_i} = E_{adi} + E_{li} \tag{29}$$

OECD (2010) and Sasaki (2012) estimated emissions from atmospheric deposition as

$$E_{adi} = \frac{1}{1000} \times 0.01 \times RF \times x_i \times \frac{44}{28} \tag{30}$$

where 0.01 is the emissions factor (kgN_2O-N/kgN), RF the rate of deposition from fertiliser (chemical fertiliser: 0.1; organic fertiliser: 0.2), and x_i N application. In this study, the estimated values of E_{adi} and x_i again depend on farm type j.

Emissions from N leaching and runoff (E_{li}) were defined by OECD (2010) and Sasaki (2012) as

$$E_{li} = \frac{1}{1000} \times 0.0124 \times Z_i(x_i) \times \frac{44}{28} \tag{31}$$

where 0.0124 is the emissions factor from N leaching and runoff (kgN_2O-N/kgN) and z_i the runoff amount (kgN). The estimated values of E_{li}, x_i, and z_i also depend on farm type j.

3.4.3 CO_2 emissions and sequestration

Lastly, the amount of carbon sequestration (tonne) is expressed as follows:

$$CO_{2i} = \sum Seq_i \times \frac{44}{12} \tag{32}$$

OECD (2010) and Sasaki (2012) estimated the functions for paddy and upland fields by using data from MAFFSTAT as follows:

$$Seq_1 = 0.0062x_{oi}^2 + 0.052x_{oi} \tag{33}$$

$$Seq_2 = 0.0013x_{oi}^2 + 0.022x_{oi} \tag{34}$$

Similarly, the estimated values of Seq_1, Seq_2, and x_{0i} depend on farm type j.

3.5 Social welfare function

The monetary valuation of environmental impacts is used to aggregate each environmental effect and then this is combined with the profit function. The social welfare function for each farm type can be expressed as

$$SW = \int_0^1 \sum \pi^{ij} - \alpha z_{1j} - \beta z_{2j} - \gamma GHG_{ij} \quad for\ i = 1,2, j = 1,2,3,4,5 \tag{35}$$

$$\text{where } \alpha = \begin{cases} 674, & z_{ij} > 0 \\ 6563, & z_{ij} < 0 \end{cases}, \quad \beta = 674, \quad \gamma = 7039$$

where π^{ij} refers to the farmer's profit function, z_{ij} to the amount of N runoff (purification), and GHG_{ij} to total GHG emissions.

OECD (2010) and Sasaki (2012) estimated the monetary valuation of N purification and runoff as 6563 and 674 JPY/kg, respectively based on Shiratani et al. (2004, 2008), who used the replacement cost method. Although other monetary valuation methods can be applied, stated preference methods (contingent valuation and choice experiment) are difficult to apply because of the unfamiliarity of N runoff and purification, which might lead to an unsuitable valuation (Hanley et al., 1997). Moreover, no such precise calculation exists in Japan. Further, these studies estimated the monetary valuation of GHG emissions as 7039 JPY/Ct based on Barker et al. (2007), which compared modelling estimations of the costs of meeting the Kyoto targets.

4 Reference level framework for the policy simulations

Although consumers expect farmers to adopt good farming practices and demand environmentally friendly agricultural products, they do not necessarily bear the associated costs. However, without economic incentives, farmers may not adopt these practices because they incur additional costs. Policymakers must thus consider the cost-effectiveness of agri-environmental policies, especially which party should bear the costs of achieving environmental targets. For this purpose, OECD (2001) developed the reference framework (Figure 1) for examining the impacts of agri-environmental policies.

Environmental reference levels are defined as the minimum level of environmental quality that farmers are obliged to provide at their own expense. Environmental targets are defined as the desired (voluntary) levels of environmental quality that go beyond the minimum requirements for the agricultural sector in a given country (OECD, 2001, 2010). If agricultural activities provide agri-environmental public goods above the reference level, this provides benefits for which farmers or landowners may need to be compensated. When agricultural activities push the level of environmental services below the reference level, farmers are required to restore the reference level at their own expense (the Polluter Pays Principle) (OECD, 1997).

Figure 1 illustrates the relationship between environmental targets and reference levels for three cases (where X represents environmental targets [X^T], reference levels [X^R], and current farming practices [X^C]). Cases A to C represent identical environmental outcomes and allocations of farm resources. What differs among them is the distribution of costs associated with achieving the defined environmental target (OECD, 2001, 2010):

- ***In Case A,*** current farming practices (X^C) provide an environmental performance below the reference level (X^R). In this case, farmers are emitting pollution ($X^C<X^R$), and they must adopt farming practices to achieve X^T *at their own expense.* If not, the government may charge a penalty to induce compliance.
- ***In Case B,*** current farming practices achieve an environmental performance corresponding to the chosen reference level ($X^C=X^R$), which is below X^T. In this case, farmers *may need to be compensated* for changing from X^C to those practices required to achieve X^T.
- ***In Case C,*** current farming practices (X^C) provide an environmental performance below X^T, but with X^R above the environmental performance level of X^C and below X^T. To improve their environmental performance, farmers must adopt appropriate farming practices *at their own expense* up to X^R otherwise the government may charge a penalty. Requirements for farmers to improve their environmental performance beyond X^R to reach X^T may also need to be remunerated.

Despite the usefulness of this reference framework, the environmental impacts of agri-environmental policies have rarely been modelled. Therefore, following the framework in Figure 1, this study carried out three policy simulations. In all three, the environmental target was set as half the current chemical fertiliser application in line with the definition of agricultural products produced in an environmentally friendly way in Japan (MAFF, 2007). Moreover, payments were set to half the application of chemical fertiliser currently used by average farming household compared with the private optimum (198 kg). Thus,

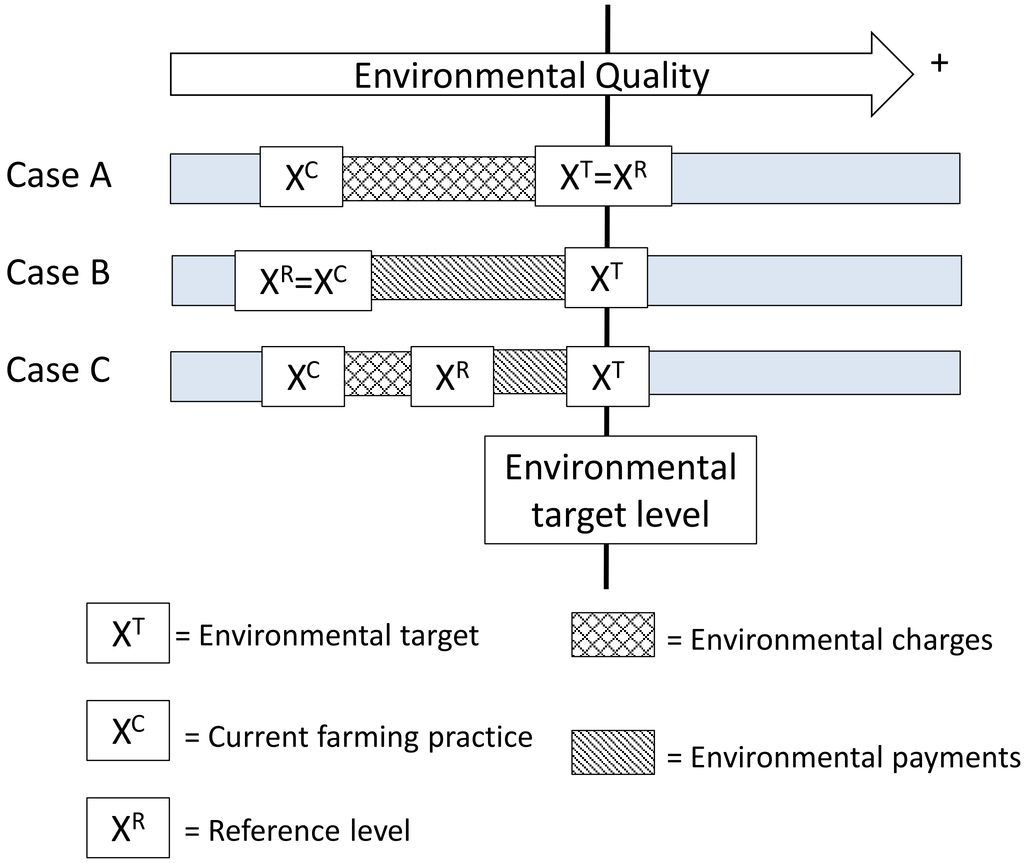

Figure 1. Environmental targets and reference levels

Source: Adapted from Uetake (2013) (originally from OECD (2001)).

- **Simulation A** (corresponding to Case A): A 50% reduction in the application of chemical fertiliser
- **Simulation B** (corresponding to Case B): An environmental payment (900 JPY per N kg reduction) for achieving the environmental target
- **Simulation C** (corresponding to Case C): A policy mix of regulation and a payment, namely a 25% reduction in the application of chemical fertiliser plus an environmental payment (900 JPY per N kg reduction) for achieving the environmental target

5 Policy simulations and results

The model described herein estimated crop production, N runoff, GHG emissions, budget outlays, and social welfare under these three scenarios. Although Japan has a rice production adjustment programme and sets a rice production quota to prevent a drop in the rice price, these external factors were excluded from our model. To reflect this impact, the rice price was rather assumed to decrease by 4.66%, which was estimated from data provided by OECD (2009). In addition, as prior conditions, Japanese direct payments for wheat production (non-environmental payments, 6360 JPY/60 kg) were incorporated into the model to reflect farmers' actual situations. Compared with the rice price (11,660 JPY/60 kg), the wheat price is much cheaper (3033 JPY/60 kg). Without payments for wheat production, farmers hardly produce wheat in Japan.

5.1 Private and social optimum

The first analysis compares the private and social optima. Farmers maximise their profit by ignoring both positive and negative externalities (private optimum), whereas government planners maximise both farmers and society's profit by incorporating both positive and negative externalities (social optimum).

Table 5 shows the land use allocation and total production of rice and wheat. Under the social optimum, farmers produce only rice except non-business farm household in mountainous areas, since rice paddy has a nitrogen purification function and offers larger benefits to society. Under the private optimum, rice paddy is not cultivated at all owing to the relatively high cost of rice production as well as limited nitrogen response characteristics even on high-quality land. Wheat is mainly cultivated on parcels in non-mountainous areas by business farm household, because it has a relatively high nitrogen response on high-quality land. In addition, this simulation allocated some land abandonment for non-business farm household in mountainous areas because of the prevailing high production costs. More land is abandoned under the social optimum because of high N runoff rates owing to the steep slopes. The areas of abandoned land, rice, and upland crops in 2010–11 were approximately 0.19 million ha (8%), 1.53 million ha (65%), and 0.62 million ha (27%), respectively. The share of the simulation results of each (abandoned land 2%, rice 72%, wheat 25%) were reasonably close to actual usage, suggesting that the results represent the reality of current land use.

Table 5.
Private and social optimums: Comparing land use and total production

	Farmer	Land use (parcel)			Total production	
		Abandoned land	Rice	Wheat	Rice (kg)	Wheat (kg)
Private optimum	BFH-N-M	0	44	17	23 847	10 431
	BFH-M	0	34	9	18 567	5 222
	N-BFH-N-M	0	13	0	7 263	0
	N-BFH-M	3	0	6	0	2 150
	AVERAGE	0	21	0	11 733	0
Social optimum	BFH-N-M	0	61	0	34 749	0
	BFH-M	0	43	0	24 488	0
	N-BFH-N-M	0	13	0	7 403	0
	N-BFH-M	6	0	3	0	1 155
	AVERAGE	0	21	0	11 959	0

Source: Authors' calculation.

Farmers maximise yields by using relatively larger amounts of chemical fertiliser under the private optimum than under the social optimum. In the latter, more organic fertiliser is used (Table 6). Under the private optimum, the application of chemical fertiliser leads to N runoff, but not for business farm household in mountainous areas and non-business farm household in non-mountainous areas because they do not produce wheat, only rice. Even under the social optimum, non-business farm household in mountainous areas generate N runoff due to wheat production, although other farm types do not (owing to the N purification function of rice paddy; Table 7). The impact of CH_4 emissions and carbon sequestration through the application of organic fertiliser on GHG emissions is fairly limited under the private optimum. By comparison, organic fertiliser has partially been substituted for chemical fertiliser under the social optimum, which significantly decreases net GHG emissions (Tables 6 and 7).

Table 6.
Private and social optimums: Comparing fertiliser use

	Farmer	Fertiliser use (total)					
		Rice		Wheat		Total	Total Organic
		Chemical (kg)	Organic (t)	Chemical (kg)	Organic (t)	Chemical (kg)	(t)
Private optimum	BFH-N-M	418	11	249	13	667	24
	BFH-M	322	9	133	7	455	15
	N-BFH-N-M	122	4	0	0	122	4
	N-BFH-M	0	0	87	2	87	2
	AVERAGE	198	6	0	0	198	6
Social optimum	BFH-N-M	404	46	0	0	404	46
	BFH-M	285	33	0	0	285	33
	N-BFH-N-M	86	10	0	0	86	10
	N-BFH-M	0	0	26	3	26	3
	AVERAGE	139	16	0	0	139	16

Source: Authors' calculation.

Table 7.
Private and social optimums: Comparing environmental effects

	Farmer	Nitrogen[1]	GHG emission and sequestration[1]			
		N runoff (purification) (kg)	CH_4(t)	N_2O (t)	CO_2 (t)	Total (t)
Private optimum	BFH-N-M	74.65	1.43	2.89	-3.05	1.26
	BFH-M	33.06	1.11	1.78	-2.14	0.75
	N-BFH-N-M	-8.36	0.43	0.31	-0.68	0.06
	N-BFH-M	39.88	0.00	0.60	-0.14	0.45
	AVERAGE	-13.51	0.70	0.50	-1.09	0.10
Social optimum	BFH-N-M	-61.28	2.54	1.61	-8.03	-3.87
	BFH-M	-43.20	1.79	1.14	-5.66	-2.73
	N-BFH-N-M	-13.06	0.54	0.34	-1.71	-0.82
	N-BFH-M	8.97	0.00	0.22	-0.20	0.02
	AVERAGE	-21.10	0.88	0.56	-2.76	-1.33

[1]. The minus represents the purification for nitrogen and the sequestration for carbon.
Source: Authors' calculation.

Under the private optimum, social welfare is always lower than that under the social optimum. Consequently, a farmer's profit in this case is lower than the profit of a farmer under the private optimum (Table 8). Non-business farm household in mountainous areas lose the most profits compared with the private optimum, since some parcels are better to be abandoned because of their low productivity and high negative environmental impacts owing to steep slopes.

Table 8.
Private and social optimums: Comparing private profits and social welfare

	Farmer	Private profits (1000 JPY)	Profits/ Private Optimum	Runoff damage (purification benefit)[1]	GHG emission damage (Sequestration benefit)[1]	Social welfare (1000 JPY)	Social welfare/ Social Optimum
Private optimum	BFH-N-M	2 262	1.00	0.4	9	2 252	0.88
	BFH-M	1 149	1.00	-44	5	1 188	0.85
	N-BFH-N-M	262	1.00	-55	0.4	316	0.92
	N-BFH-M	38	1.00	71	3	-36	-4.18
	AVERAGE	573	1.00	-89	1	661	0.94
Social optimum	BFH-N-M	2 144	0.95	-402	-27	2 573	1.00
	BFH-M	1 093	0.95	-284	-19	1 396	1.00
	N-BFH-N-M	251	0.96	-86	-6	343	1.00
	N-BFH-M	25	0.64	16	0.1	8	1.00
	AVERAGE	556	0.97	-138	-9	704	1.00

1. The minus represents the purification for nitrogen and the sequestration for carbon.
Source: Authors' calculation.

5.2 Agri-environmental policy simulations

The second analysis compares the three above-mentioned policy options for reducing N runoff and GHG emissions. These policy scenarios, which assume profit maximisation by farmers, are regulation, environmental payment, and a combination of both. Organic fertiliser is substituted for chemical fertiliser substantially compared with the private optimum in all three simulations (Tables 6 and 9).

Table 9.
Agri-environmental policy simulations: Comparing fertiliser use

	Farmer	Fertiliser use (total)					
		Rice (kg)		Wheat (kg)		Total Chemical (kg)	Total Organic (t)
		Chemical	Organic	Chemical	Organic		
Regulation (N 50% reduction)	BFH-N-M	237	48	81	16	317	65
	BFH-M	184	38	29	6	214	44
	N-BFH-N-M	61	13	0	0	61	13
	N-BFH-M	0	0	37	4	37	4
	AVERAGE	99	21	0	0	99	21
Payment (900JPY/ N1kg reduction)	BFH-N-M	205	41	185	21	389	62
	BFH-M	158	31	100	12	258	43
	N-BFH-N-M	64	12	0	0	64	12
	N-BFH-M	0	0	28	7	28	7
	AVERAGE	103	20	0	0	103	20
Policy mix (N 25% reduction + 900JPY/ N1kg reduction)	BFH-N-M	220	44	181	15	401	59
	BFH-M	174	34	71	8	244	42
	N-BFH-N-M	64	12	0	0	64	12
	N-BFH-M	0	0	28	6	28	6
	AVERAGE	103	20	0	0	103	20

Source: Authors' calculation.

Table 10 summarises the land use allocation and total production of rice and wheat under the three simulations. Business farm household increase rice production compared with the private optimum under the regulation policy, because wheat production is more responsive to N application than rice is. Halving the N application significantly reduces wheat yields so that producing rice become more profitable on some parcels. On the contrary, the environmental payment policy allocates more land to wheat because the payment is based on reducing nitrogen compared with the amount under the private optimum (where farmers use more nitrogen for wheat than for rice). Therefore, if an environmental payment were introduced, farmers would receive larger payments by producing wheat than by producing rice. Thus, farmers would produce more wheat. Finally, the policy mix allocates more parcels to rice compared with the payment policy, but fewer parcels versus the regulation policy.

Table 10.
Agri-environmental policy simulations: Comparing land use and total production

	Farmer	Land use (parcel)			Total production	
		Abandoned land	Rice	Wheat	Rice (kg)	Wheat (kg)
Regulation (N 50% reduction)	BFH-N-M	0	50	11	27 859	6 742
	BFH-M	0	39	4	21 934	2 328
	N-BFH-N-M	0	13	0	7 384	0
	N-BFH-M	4	0	5	0	1 798
	AVERAGE	0	21	0	11 929	0
Payment (900JPY/ N1kg reduction)	BFH-N-M	0	43	18	23 659	10 963
	BFH-M	0	33	10	18 279	5 774
	N-BFH-N-M	0	13	0	7 385	0
	N-BFH-M	0	0	9	0	2 740
	AVERAGE	0	21	0	11 930	0
Policy mix (N 25% reduction + 900JPY/ N1kg reduction)	BFH-N-M	0	46	15	25 447	9 317
	BFH-M	0	36	7	20 094	4 097
	N-BFH-N-M	0	13	0	7 385	0
	N-BFH-M	2	0	7	0	2 288
	AVERAGE	0	21	0	11 930	0

Source: Authors' calculation.

The environmental impacts also depend on policy measures and farm types (Table 11). Because of the reduced application of chemical fertiliser, there is a significant reduction in N runoff and increase in N purification compared with the private optimum (Table 7) for the regulation and policy mix measures. However, the payment for N reduction provides farmers with an incentive to allocate more parcels to wheat production or cultivate underused parcels under the private optimum, which increases N runoff for business farm household and non-business farm household in mountainous areas. Regulation reduces the N runoff the most among the three policy simulations, while the increased application of organic fertiliser raises CH_4 emissions in all cases. However, reduced N_2O and increased carbon sequestration through the application of organic fertiliser significantly decreases net GHG emissions compared with emissions under the private optimum for all simulations and all farm types.

Table 11.

Agri-environmental policy simulations: Comparing environmental effects

	Farmer	Nitrogen[1]	GHG emission and sequestration[1]			
		N runoff (purification) (kg)	CH_4(t)	N_2O (t)	CO_2 (t)	Total (t)
Regulation (N 50% reduction)	BFH-N-M	-25.59	2.25	2.23	-9.34	-4.86
	BFH-M	-31.85	1.76	1.36	-6.84	-3.72
	N-BFH-N-M	-14.10	0.59	0.35	-2.16	-1.22
	N-BFH-M	-13.03	0.00	0.32	-0.32	0.01
	AVERAGE	-22.78	0.95	0.56	-3.49	-1.98
Payment (900JPY/ N1kg reduction)	BFH-N-M	18.85	1.92	2.89	-8.47	-3.65
	BFH-M	39.83	1.48	2.00	-6.16	-2.68
	N-BFH-N-M	-8.36	0.58	0.37	-2.11	-1.15
	N-BFH-M	67.77	0.00	0.74	-0.51	0.23
	AVERAGE	-22.66	0.94	0.56	-3.40	-1. 90
Policy mix (N 25% reduction + 900JPY/ N1kg reduction)	BFH-N-M	17.23	2.06	2.66	-8.53	-3.80
	BFH-M	-13.98	1.61	1.61	-6.40	-3.18
	N-BFH-N-M	-14.03	0.58	0.34	-2.11	-1.18
	N-BFH-M	14.72	0.00	0.38	-0.48	-0.09
	AVERAGE	-22.66	0.94	0.56	-3.40	-1.90

1. The minus represents the purification for nitrogen and the sequestration for carbon.
Source: Authors' calculation.

Table 12 summarises the profit and social welfare under the three simulations. Regulation improves social welfare the most compared with the private optimum. However, farmers' profits decrease under this scenario even though they create environmental benefits for society through N purification and GHG sequestration (see Table 11). By contrast, the environmental payment increases farmers' profits significantly but does not necessarily improve social welfare because of the large budgetary costs of the policy and land allocation shifts to wheat. For business farm household in mountainous areas and non-business farm household in mountainous and non-mountainous areas, social welfare even worsens compared with under the private optimum. Lastly, the policy mix improves social welfare compared with the private optimum without reducing farmers' profits.

Table 12.

Agri-environmental policy simulations: Comparing private profits and social welfare

	Farmer	Profits (1000 JPY)	Profit+ payment (1000 JPY)	Profits/ Private optimum	Budget outlays (1000 JPY)	Runoff damage[1] (purification benefit)	GHG emission damage[1] (Sequestration benefit)	Social welfare (1000 JPY)	Social welfare/ Social optimum
Regulation (N 50% reduction)	BFH-N-M	2 082	2 082	0.92	0	-305	-34	2 421	0.94
	BFH-M	1 035	1 035	0.90	0	-259	-26	1 321	0.95
	N-BFH-N-M	233	233	0.89	0	-93	-9	334	0.97
	N-BFH-M	25	25	0.66	0	23	-0.1	2	0.25
	AVERAGE	526	526	0.92	0	-149	-14	689	0.98
Payment (900JPY/ N1kg reduction)	BFH-N-M	2 134	2 389	1.06	255	-190	-26	2 349	0.91
	BFH-M	1 081	1 263	1.10	182	-29	-19	1 105	0.79
	N-BFH-N-M	235	288	1.10	53	-55	-8	298	0.87
	N-BFH-M	-12	72	1.89	85	120	2	-134	-15.79
	AVERAGE	527	615	1.07	89	-149	-13	689	0.98
Policy mix (N 25% reduction + 900JPY/ N1kg reduction)	BFH-N-M	2 159	2 241	0.99	82	-208	-27	2 394	0.93
	BFH-M	1 058	1 138	0.99	80	-211	-22	1 292	0.93
	N-BFH-N-M	235	260	1.00	25	-92	-8	335	0.98
	N-BFH-M	4	47	1.23	43	26	-1	-21	-2.51
	AVERAGE	530	571	1.00	41	-149	-13	692	0.98

1. The minus represents the purification for nitrogen and the sequestration for carbon.
Source: Authors' calculation.

In summary, there are significant differences among the three policy scenarios in terms of cost burdens. Under the regulation policy, since all the costs for achieving the environmental target (50% reduction in N application) are borne by farmers (Case A), farmers' profits are the lowest, but there are no budget outlays. By contrast, under the environmental payment policy, since farmers are remunerated for producing environmental benefits (Case B), their profits are highest with the highest budget outlays, which are covered by taxpayers. Thus, the costs for achieving environmental targets are now transferred from farmers to taxpayers. The policy mix of regulation and payment asks both farmers and taxpayers to bear the cost of achieving the environmental target (Case C). Therefore, farmers' profits are larger than they are in the case of the regulation, but lower than they are in the case of the payment; moreover, budget outlays are smaller than they are under the payment policy, but larger than they are under the regulation policy.

Depending on farm type, land use allocation, production, and environmental externalities also vary. For average farmers, all the policy simulations improve social welfare (0.98) compared with the private optimum (0.94). However, these three policy measures do not necessarily improve social welfare for all farmers. In particular, non-business farm household in mountainous areas create negative social impacts because of their high production costs and the high N runoff associated with land on steep slopes. Indeed, even after the introduction of the environmental payment and the policy mix, social welfare remains negative. These farm-related environmental impacts suggest that more targeted approaches are necessary rather than developing agri-environmental policies that target the average farmer.

The presented simulation results suggest that the policy mix is the preferred measure in terms of economics, benefits to the environment, and equity and cost sharing. However, although farmers are producing environmental benefits, their private profits do not increase compared with the private optimum. The question thus remains of whether remuneration and incentives for farmers who adopt good farming practices is enough. One way in which to design a better policy mix would be to develop a measure that targets business farm household. This targeted policy may make it possible to reallocate the budget used for non-business farm household to business farm household, increasing their profits under the current budget ceiling as well as improving the environment and social welfare through the reduction of costly farming by non-business farm household, especially in mountainous areas.

6 Discussion and conclusion

This study used the SAPIM in order to analyse the degree to which agricultural policies address the increasing consumer demand for environmentally friendly products in Japan. In particular, it identified the environmental effects of different agricultural policies based on farms' characteristics. Three agri-environmental policy simulations were selected based on the OECD reference level framework, the results of which indicate that these policies affect land use, production, and environmental externalities. According to the simulation results, a policy mix of regulation and an environmental payment reduces the impact on the environment and keeps farmers' profits high.

This study was based on a set of key assumptions relating to output prices, input prices, and the valuation of environmental externalities. The key source of uncertainty is arguably associated with the valuation of social benefits. To test the sensitivity of the presented results, we followed Pannell (1997) to conduct a sensitivity analysis by changing the monetary value of N runoff (purification) and GHG emissions by 10% and 30%. We found that all the key findings of this study continue to hold within the range of the sensitivity analysis, suggesting that even a 30% shock does not change the results obtained fundamentally.

This integrated model approach is subject to limitations with respect to the data, model parameters, and economic and biophysical relationships. However, this approach remains a valuable tool for enabling policymakers to design and implement effective and efficient policies and move towards a sustainable food system.

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