ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ

ᑲᑎᒪᔅᔪᑎᒃᓴᑦ: ᑲᑎᒪᕕᒃᓴᖓᒍᑦ ᑲᑎᒪᓂᖅ 003-2024

ᐊᒃᑑᕙ 30, 2024

ᐊᕐᕕᐊᑦ, ᓄᓇᕗᑦ

	ᓈᓴᐅᑦ	ᑲᑎᒪᔅᔪᑎᒃᓴᖅ	ᓇᓂᑦᑎ	ᐅᖃᓚᒪᔪᖅ:	ᐊᑯᓂᐅᓂᖓ:
9:00 - 9:02 AM	1	ᒪᑐᐃᖅᓯᓂᖅ ᑲᑎᒪᓂᕐᒥᒃ		ᐃᒃᓯᕙᐅᑕᖅ	2 ᒥᓂᑦᔅ
9:02 - 9:04 AM	2	ᑲᑎᒪᔅᔪᑎᒃᓴᓄᑦ ᐊᒡᕕᐊᕈᑎᖃᕐᓂᖅ		ᐃᒃᓯᕙᐅᑕᖅ	2 ᒥᓂᑦᔅ
9:04 - 9:05 AM	3	ᑲᑎᒪᔅᔪᑎᒃᓴᑦ: ᕿᒥᕐᕈᔭᐅᓂᖏᑦ ᐊᖏᖅᑕᐅᓂᖏᓪᓗ ᑲᑎᒪᓂᖅ RM003-2024	1	ᐃᒃᓯᕙᐅᑕᖅ	1 ᒥᓂᑦ
9:05 - 10:00 AM	4	ᖃᐅᔨᔭᐅᔪᑦ 2021-ᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᐊᒻᒪ ᓄᓇᓕᖕᓂ ᑐᓴᕋᓱᐊᕈᑎᖃᓚᐅᖅᑎᓪᓗᒋᑦ ᐱᖓᖕᓇᖓᓄᐊᖅᑕᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓇᓄᐃᑦ	2	ᓄᓇᕗᑦ ᑲᕙᒪᖓ - ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᕕᖓᑦ	
10:00 - 10:15 AM		**ᕿᑲᑲᐃᓐᓇᖅ**			15 ᒥᓂᑦᔅ
10:15 - 12:00 PM	4	ᖃᐅᔨᔭᐅᔪᑦ 2021-ᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᐊᒻᒪ ᓄᓇᓕᖕᓂ ᑐᓴᕋᓱᐊᕈᑎᖃᓚᐅᖅᑎᓪᓗᒋᑦ ᐱᖓᖕᓇᖓᓄᐊᖅᑕᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓇᓄᐃᑦ	2	ᓄᓇᕗᑦ ᑲᕙᒪᖓ - ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᕕᖓᑦ	
		ᐅᓪᓗᑭᒻᒥᓴᕐᓇᖅ			
1:15 - 3:00 PM	5	ᖃᐅᔨᔭᐅᔪᑦ 2021-ᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᐊᒻᒪ ᓄᓇᓕᖕᓂ ᑐᓴᕋᓱᐊᕈᑎᖃᓚᐅᖅᑎᓪᓗᒋᑦ ᓂᒋᐊᓄᐊᖅᑕᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓇᓄᐃᑦ	3	ᓄᓇᕗᑦ ᑲᕙᒪᖓ - ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᕕᖓᑦ	
		ᕿᑲᑲᐃᓐᓇᖅ			15 ᒥᓂᑦᔅ
3:15 - 5:00 PM	6	ᖃᒪᓂᕐᔪᐊᑉ ᑐᒃᑐᖏᑕ ᐅᓄᕐᓂᖏᑦ ᐊᒻᒪ ᑐᒃᑐᓕᕆᔅᔪᑏᑦ ᐊᐅᓚᑦᑎᔅᔪᑏᑦ	4	ᓄᓇᕗᑦ ᑲᕙᒪᖓ - ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᕕᖓᑦ	
	7	ᓄᖅᑲᕐᓂᖅ ᑲᑎᒪᓂᐅᑉ RM003-2024			

ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ

ᑲᑎᒪᔾᔪᑎᒃᓴᑦ: ᑲᑎᒪᕕᒃᓴᖓᒍᑦ ᑲᑎᒪᓂᖅ 003-2024

ᐊᒃᑑᕙ 31, 2024

ᐊᕐᕕᐊᑦ, ᓄᓇᕗᑦ

	ᓈᓴᐅᑦ	ᑲᑎᒪᔾᔪᑎᒃᓴᖅ	ᓇᓂᑦᑎ	ᐅᖃᓕᒪᔪᖅ:	ᐊᑯᓂᐅᓂᖓ:
9:00 - 9:02 AM	1	ᒪᑐᐃᖅᓯᓂᖅ ᑲᑎᒪᓂᕐᒥᒃ		ᐃᒃᓯᕙᐅᑕᖅ	2 ᒥᓂᑦᔅ
9:02 - 9:04 AM	2	ᑲᑎᒪᔾᔪᑎᒃᓴᓄᑦ ᐊᒃᕕᐊᕈᑎᖃᕐᓂᖅ		ᐃᒃᓯᕙᐅᑕᖅ	2 ᒥᓂᑦᔅ
9:04 - 9:05 AM	3	ᑲᑎᒪᔾᔪᑎᒃᓴᑦ: ᕿᒥᕐᕈᔭᐅᓂᖏᑦ ᐊᖏᖅᑕᐅᓂᖏᓪᓗ ᑲᑎᒪᓂᖅ RM003-2024	1	ᐃᒃᓯᕙᐅᑕᖅ	1 ᒥᓂᑦ
9:05 - 10:00 AM	4	ᐊᖑᔭᐅᔪᒃᓴᑦ ᐃᓗᐃᑦᑑᓈᖅᓯᒪᔪᑦ ᑰᑕᖏᑦ ᖃᓕᕋᓕᒐᒃᓴᑦ ᖃᓕᕋᓕᖕᓂᐊᕐᕕᖕᒥ 0-ᒥ ᐊᕐᕌᒍᐊᓂ 2025 ᐊᒻᒪᓗ 2026 ᖃᓕᕋᓕᖕᓂᐊᕐᓇᕐᒥ	5		
10:00 - 10:15 AM		**ᕿᑲᑲᐃᓐᓇᖅ**			15 ᒥᓂᑦᔅ
10:15 - 11:00 AM	5	ᑐᓴᖅᑎᑦᑎᔾᔪᑏᑦ ᓴᓇᔭᐅᔪᑦ ᑲᑐᔾᔭᐅᓪᓗᑎᒃ ᓄᓇᕗᒥ ᐃᖃᓪᓕᐊᕐᓂᕐᒧᑦ ᒪᓕᒐᕆᓂᐊᖅᑕᖏᑦ	6		
11:00 - 12:00 PM	6	ᐃᖅᑲᓗᓕᕆᔨᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᑦ ᑲᓇᑕᒥ ᑐᓴᖅᑎᑦᑎᔾᔪᑎᖏᑦ - ᑕᕆᐅᕐᒥ ᐃᒪᕐᒥ ᐱᐅᖅᓱᐊᕐᕕᖃᕐᓂᖅ	7		
		ᐅᓪᓗᑭᒻᒥᓴᕐᓇᖅ			
1:15 - 2:00 PM	7	ᑐᓴᖅᑎᑦᑎᔾᔪᑦ ᐸᕐᓇᐅᑎᐅᑲᐃᓐᓇᖅᑐᒥᒃ ᑕᓪᓗᕈᑎᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᐱᐅᖅᓱᐊᖅᑕᐅᓗᓂ ᓴᐳᒻᒥᐅᓯᖅᑕᐅᓯᒪᓂᖓ	8		
	8	ᓄᖅᑲᕐᓂᖅ RM003-2024			

<u>ᐅᑯᓄᖓ ᑐᓂᔭᐅᔪᖅ</u>

<u>ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ</u>

<u>ᐆᒧᖓ</u>

ᑐᓴᖅᑕᐅᔪᒃᓴᑦ: **X** **ᐃᓱᒪᓕᐅᕈᑕᐅᔪᑦ:**

ᐱᔾᔪᑕᐅᔪᖅ: 2021–ᒥᑦ ᖃᐅᔨᓴᕈᑎᕗᓂᑦ ᓄᓇᓕᖕᓂᓪᓗ ᑐᓴᕋᓱᖕᓂᖅ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᑕ ᓇᓄᖏᓐᓂᑦ.

ᐱᓕᕆᐊᖑᓯᒪᔪᑦ:

- ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᓇᓅᖃᑎᒌᑦ ᑲᑐᑎᑕᐅᔪᑦ ᒫᓂᑑᐸᒥᑦ ᓄᓇᕗᒻᒥᓪᓗ (ᐊᑦᔨᖖᒍᐊᖅ 1). ᑭᖑᓪᓕᖅᐹᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᓪᓗᑎᒃ ᐅᓄᕐᓂᖏᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓪᓗᑎᒃ ᐱᓕᕆᐊᖑᓚᐅᖅᐳᑦ 2011-ᒥ 2016-ᒥᓗ, 949 ᐊᒻᒪ 842-ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ ᓇᓄᐃᑦ, ᐊᑐᓂ. ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᓇᓅᖃᑎᒌᑦ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᐊᓇᓱᒋᔭᐅᓚᐅᖅᐳᑦ ᑕᐃᔅᓱᒪᓂ.

- ᓄᓇᓖᑦ ᐊᕐᕕᐊᓂ, ᑎᑭᕋᕐᔪᐊᕐᒥ, ᐊᒻᒪ ᑲᖏᖅᖠᓂᕐᒥ ᐊᖑᓇᓱᒃᐸᒃᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ. ᒫᓐᓇᐅᔪᖅ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ 38–ᖑᕗᑦ ᓇᓄᐃᑦ ᐊᕐᕌᒍᓕᒫᖅ. ᑲᑎᔾᔪᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐋᖅᑭᒃᑕᐅᓚᐅᖅᐳᑦ ᐊᓯᔾᔨᕆᐊᒐᒃᓴᐅᓪᓗᓂ ᑰᑕᖏᑦ (FQS) 2:1–ᒥ ᐊᖑᓇᓱᒃᑕᐅᔪᖅ ᑭᓲᓂᖏᑕ ᖃᔅᓯᐅᓂᖏᓐᓄᑦ ᑲᔪᓯᑎᑕᐅᓪᓗᓂᓗ 1:1–ᒥ ᐊᖑᔭᐅᔪᑦ ᑭᓲᓂᖏᑕ ᖃᔅᓯᐅᓂᖏᓐᓄᑦ ᓄᑖᖅ ᐊᖑᓇᓱᒃᑕᐅᔪᓄᑦ ᐊᐅᓚᑦᑎᔾᔪᑎ ᓈᓴᖅᑕᐅᔾᔪᑎᖏᓪᓗ (HACCS) ᐊᑐᓕᓚᐅᖅᑎᓪᓗᒍ 2022–ᒥ. ᓇᓐᓄᒃᑕᐅᒪᔪᒃᑐᑦ 2012/2013 ᐊᒻᒪ 2022/2023 ᐊᑯᓐᓂᖓᓂ 28–ᖑᓚᐅᖅᐳᑦ ᓇᓄᐃᑦ ᐊᕐᕌᒍᒥ ᐊᑕᐅᓯᕐᒥ (ᐊᑦᔨᖖᒍᐊᖅ 2). ᓇᓐᓄᒃᑕᐅᒪᔪᒃᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᐊᕐᕌᒍᓂᑦ ᖁᓕᓂᑦ ᐊᓂᒍᖅᑐᓂᑦ (2012/2013–ᒥᑦ 2022/2023–ᒧᑦ) 26% ᐊᑖᓃᐸᓗᓚᐅᖅᐳᖅ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᑲᑎᔾᔪᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ.

- ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᐊᒻᒪ ᓂᒋᖅᐸᓯᐊᓂ ᑲᖏᖅᓯᐊᓘᑉ (SH) ᒪᓕᖅᑦᑕᖅᓯᒪᕗᑦ ᐊᕐᕌᒍᓄᑦ ᑕᓪᓕᒪᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ, ᖃᖓᑦᑎᐊᓕᓴᐅᓲᖅ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᐱᐊᓂᒃᑕᐅᓚᐅᖅᓱᑎᒃ 2021–ᒥ. ᓄᓇᓕᖕᓂ ᑐᓴᕋᓱᑲᑕᓚᐅᖅᑎᓪᓗᒋᑦ ᐅᖃᓘᑎᒃᑯᑦ 2021–ᒥ (ᐅᐸᒍᑎᓪᓗᑎᒃ ᑐᓴᕋᓱᒍᓐᓇᓚᐅᙱᓚᑦ ᓄᕙᒡᔪᐊᕐᓇᖅ-19–ᒧᑦ ᐊᐅᓪᓚᑲᑕᑐᐊᓇᐊᖅᑲᙱᑎᓪᓗᒋᑦ), ᖃᐅᔨᓴᖅᑕᐅᓯᒋᐊᓚᐅᖅᐳᖅ ᐋᒌᓯ 2021–ᒥ ᐱᐊᓂᒃᓱᑎᓪᓗ ᓯᑎᐱᕆ 2021–ᒥ. ᖃᖓᑕᓲᒃᑯᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕈᑎᒥᒃ ᐊᑐᓚᐅᖅᐳᑦ ᑖᔅᓱᒥᖓ ᖃᐅᔨᓴᖅᑎᓪᓗᒋᑦ. ᑲᖏᖅᓯᐊᓘᑉ ᓂᒋᖅᐸᓯᐅᓂ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᖅ ᑕᐃᑲᓂᑦᑕᐃᓐᓇᖅ ᐋᓐᑎᐅᕆᐅ ᒪᕙᒪᖓᓂ ᐊᒻᒪ ᑯᐸᐃᒃ ᒪᕙᒪᖓᓂ.

- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ (ECCC) ᖃᐅᔨᓴᖃᑦᑕᖅᓯᒪᕗᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ 1980–ᒥ ᐱᒋᐊᖅᓱᓂ. ᖃᐅᔨᓴᖃᑦᑕᐃᓐᓇᖅᑐᑦ ᐃᒪᐃᓕᐅᖅᓱᑎᒃ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᓪᓗᑎᒃ ᓇᐅᒃᑲᖅᑕᐅᒻᒧᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂᒃ ᐱᓕᕆᐊᖑᓯᒪᓕᖅᑐᑦ 2017–ᒥ ᐱᒋᐊᓚᐅᖅᓯᒪᓪᓗᓂ ᒫᓂᑑᐸᒥ ᐊᒻᒪ ᐋᓐᑎᐅᕆᐅ ᓯᒃᔭᖓᓂ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᖅᑐᑦ ᓇᒧᙵᐅᕙᓪᓕᐊᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᑲᑕᓚᐅᖅᐳᑦ ᐅᑯᓇᓂ, ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ/ᑲᖏᖅᓯᐊᓘᑉ ᓂᒋᐊᓂ ᐊᒻᒪᓗ ᓯᑯᐃᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᓐᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᒥ. ᖃᐅᔨᓴᕈᑕᐅᔪᑦ ᑕᕝᕙᙵᑦ ᐅᓂᒃᑲᒥ ᑐᑭᓯᓇᑦᑎᐊᓚᐅᖅᐳᑦ ᐃᑲᔫᑕᐅᓪᓗᑎᓪᓗ ᑐᑭᓯᓇᖅᓯᑎᓪᓗᒋᑦ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᖅᓯᒪᔪᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᑦ 2021–ᒥ ᓄᓇᕗᑦ ᒪᕙᒪᒃᑯᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ.

ᒪᒃᐱᖅᑐᒐᖅ 1 6-ᖑᓪᓗᑎᒃ

ᒫᓐᓇ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ:

- ᑭᖑᓪᓕᖅᐹᖅ ᐅᓂᒃᑳᖅ ᖃᐅᔨᓴᕈᑕᐅᔪᓪᓗ 2021−ᒥ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ ᑐᓂᐅᖅᑲᐅᑕᐅᓚᐅᖅᐳᑦ ᐊᐅᓚᑦᑎᔨᐅᖃᑕᐅᔪᓕᒫᓄᑦ ᓄᕕᐱᕆ 2022−ᖑᑎᓪᓗᒍ. ᓄᑖᑦ ᐅᓄᕐᓂᖏᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᑦ 618 ᓇᓄᐃᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᓂᑦ ᐅᓄᙱᓐᓂᖅᓴᐅᕗᑦ ᑭᖑᓪᓕᖅᐸᒥ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᓂᑦ 2011 ᐊᒻᒪ 2016.
- ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂᓗ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ 2011, 2016, ᐊᒻᒪ 2021−ᒧᑦ ᑲᑎᑕᐅᓚᐅᖅᐳᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᐅᓂᒃᑳᓕᐊᒥ. ᑕᒪᓐᓇ ᓴᖅᑭᑎᑦᑎᓚᐅᖅᐳᖅ ᑲᑎᓪᓗᒋᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᓂᑦ 1892, 1622, 1621−ᓂᒡᓗ ᓇᓄᕐᓂᒃ, ᐊᑐᓂ. ᑕᒪᕐᒥᒃ ᓇᓅᖃᑎᒌᑦ ᑲᑎᓯᒪᑎᓪᓗᒋᑦ, ᑕᑯᒃᓴᐅᑎᑦᑎᕗᖅ ᓱᕐᕌᓗᐊᙱᑦᑐᓂᒃ ᐅᓄᕐᓂᖏᓐᓂᒃ (ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᐊᒻᒪ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂ) 2016 ᐊᒻᒪ 2021 ᐊᑯᓐᓂᖓᓂ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ.
- ᑭᒡᓕᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᐃᕙᒃᑐᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ, ᓇᓗᓇᓚᐅᖅᑐᖅ ᖃᓄᑎᒋ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᖅᑎᒋᖕᒪᖔᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ 2021−ᒥᑦ ᓇᓄᐃᑦ ᓅᐸᓪᓕᐊᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂ, ᐊᕙᑎᓕᕆᔾᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᐋᓐᑎᐅᕆᐅᒥᓪᓗ ᓇᑭᙶᖕᓂᖏᓐᓂᑦ ᓇᓗᓇᐃᒃᑯᑦᑎᖅᓯᐃᓃᑦ ᐱᔭᐅᒃᑲᓐᓂᖅᐸᒃᑐᓪᓗ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᖅᑕᐅᓪᓗᑎᒃ.
- ᐊᕙᑎᓕᕆᔾᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᑲᑕᒃᑐᑦ ᐅᓂᒃᑳᖓᓂ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᓚᐅᖅᐳᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ 2021−ᖑᑎᓪᓗᒍ ᓇᓄᐃᑦ ᓅᐸᓪᓕᐊᓂᖏᓐᓄᑦ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂ, ᓯᑯᖃᕐᓂᖓᓄᑦ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᓗᒍ.
- ᐊᕙᑎᓕᕆᔾᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᑲᑕᒃᑐᑦ ᐅᓂᒃᑳᖓᓂ ᖃᐅᔨᓴᓚᐅᕐᒥᔪᑦ ᓇᓐᓄᒡᕕᐅᔪᕕᓂᕐᓂ. ᓇᓗᓇᐃᖅᓯᒃᑲᓐᓂᖅᐳᖅ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᐊᐅᓚᑦᑎᔨᑦ ᑭᒡᓕᒋᔭᖓ ᐊᑐᕈᓐᓇᕐᓂᖓᓂᒃ ᓇᓄᐃᑦ ᐊᖑᔭᐅᔪᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᓯᕗᓪᓕᕐᒥ ᑎᒥᖓᓂ ᐲᖅᓯᕚᐅᓚᐅᕐᓂᖏᓐᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ, ᓇᓄᐃᑦ ᐊᖑᔭᐅᔪᑦ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂᓕ ᓇᓐᓄᒃᑕᐅᔪᑦ ᓯᕗᓪᓕᖅᐸᒥ ᑎᒥᖓᓂ ᐲᖅᓯᕚᐅᓚᐅᖅᓱᑎᒃ ᑲᖏᖅᓱᐊᓘᑉ ᓂᒋᐊᓂ.

ᑐᓴᕋᓱᒡᕕᐅᔪᑦ:

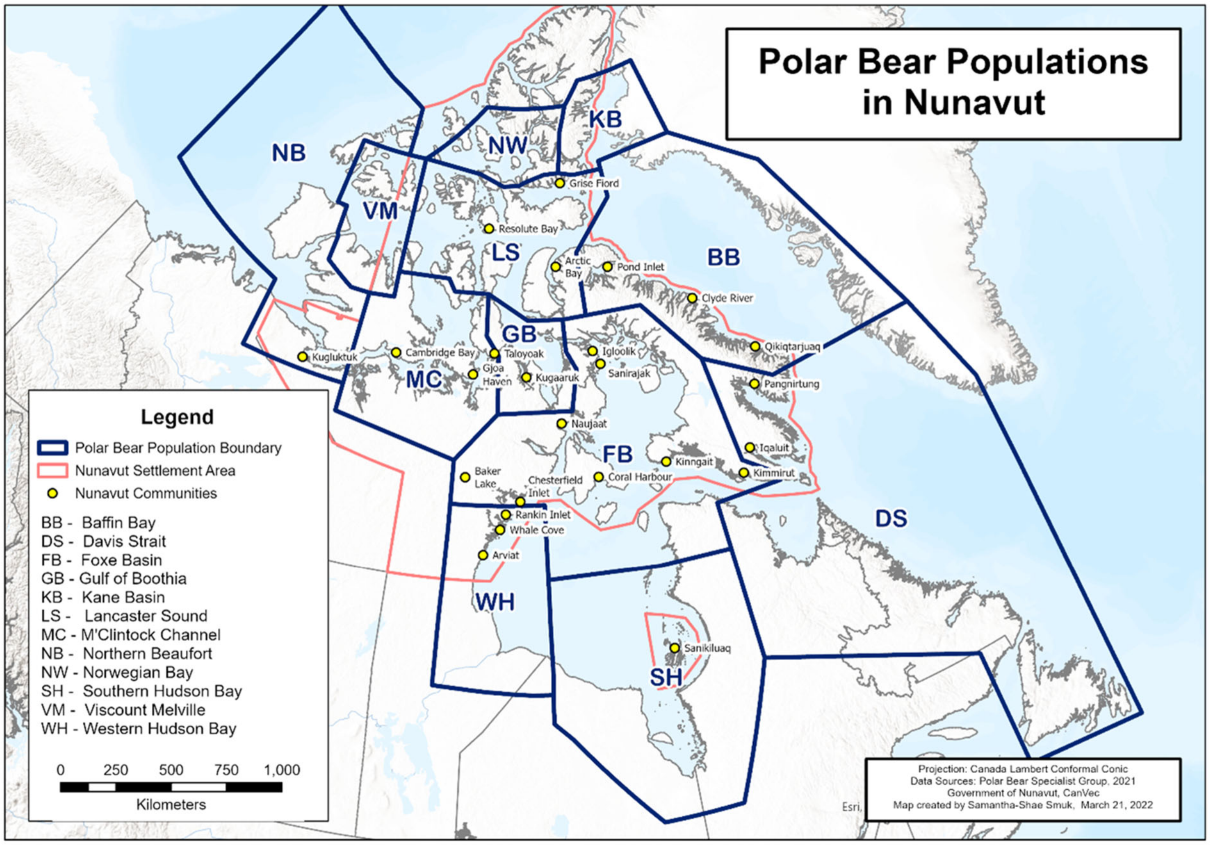
ᐅᐸᒃᓱᑎᒃ ᓄᓇᓕᖕᓂᑦ ᑐᓴᕋᓱᓚᐅᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᑕ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᖏᓐᓂ ᒫᑦᓯ 26−27−ᓗ ᒪᐃ 9−10−ᓗ ᐊᑯᓐᓂᖓᓂ, 2024. ᑐᓴᕋᓱᒃᑎᓪᓗᒋᑦ ᐱᔾᔪᑎᖃᓗᐊᓚᐅᖅᐳᑦ ᖃᐅᔨᓴᖅᑕᐅᔾᔪᑎᓂᒃ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᓈᓴᐃᑎᓪᓗᒋᑦ ᐊᒻᒪᓗ ᐊᕙᑎᓕᕆᔾᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖓᑕ ᐲᖅᓯᓪᓗᑎᒃ ᖃᐅᔨᓴᖅᑐᓂᑦ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᓱᒫᓘᑎᖃᓚᐅᖅᑐᑦ ᓈᒻᒪᒃᑲᓗᐊᕐᒪᖔᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᔪᑦ ᑐᓴᕈᒪᓚᐅᖅᓱᑎᒡᓗ ᖃᐅᔨᓴᖅᑕᐅᔾᔪᑎᖏᓐᓂ ᑎᒥᖓᑕ ᐲᖅᓯᕚᐅᓪᓗᑎᒃ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᓄᖅᓯᒋᐊᖅᑎᑦᑎᔪᒪᓚᐅᖅᑐᑦ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᑭᒃᑯᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᕐᓂᖓ ᐃᓱᒫᓘᑎᖃᕐᓂᖏᓐᓂ ᐃᓕᖅᑯᓯᕐᒥᓂᒡᓗ ᓴᐳᒻᒥᔭᐅᓗᑎᒃ. ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖏᑦ ᐊᒻᒪ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᑲᑎᒪᖃᑕᐅᖃᑦᑕᓚᐅᖅᑐᑦ ᖃᔾᔨᖅᓯᒃᑲᓪᓚᒃᓱᑎᒃ (ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᑕᑯᓗᒋᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᑲᑎᒪᑎᓪᓗᒋᑦ ᓇᐃᓈᖅᓯᒪᔪᑦ ᐅᓂᒃᑳᖏᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔾᒃᑯᓐᓂ).

ᐊᑐᓕᖅᑯᔭᐅᔪᑦ:

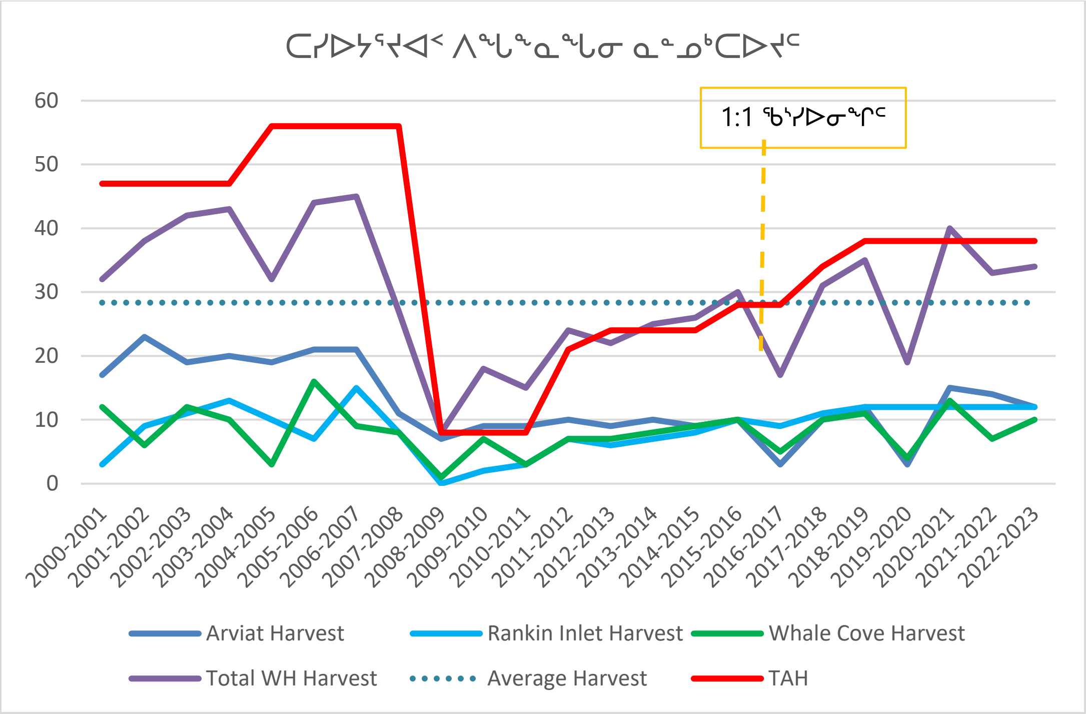
1. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ **ᐊᓯᔾᔩᖁᔨᙱᑦᑐᑦ ᒫᓐᓇ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ 38–ᖑᓪᓗᑎᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᓇᓅᖃᑎᒌᑦ**. ᐃᓱᒪᒋᓪᓗᒋᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᒻᒪ ᐱᖓᖕᓇᖅᐸᓯᐊᓂ ᓇᓚᐅᑦᑖᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᖦᖢᒋᑦ 2016 ᐊᒻᒪ 2021 ᐊᑯᓐᓂᖓᓂ, ᓇᓄᐃᑦ ᓱᕐᕌᓗᐊᖅᓯᒪᙱᑦᑐᑦ ᒫᓐᓇ ᐊᕙᑎᖏᑦ ᖃᓄᐃᓕᖓᓂᖏ ᐊᖑᓇᓱᒃᑕᐅᕙᒃᑐᓪᓗ ᒪᓕᒃᖢᒋᑦ.

2. ᒫᓐᓇ ᑲᓇᑕᓕᒫᒥ ᓄᓇᕐᔭᐊᓕᒫᒥᓗ ᓇᓄᐃᑦ ᐊᐅᓚᑕᐅᔪᑦ ᐱᓕᕆᐊᖑᕙᓪᓕᐊᔪᑦ ᒪᓕᒡᓗᒍ, ᐊᕙᑎᔨᕆᔨᒃᑯᑦ ᑐᒃᓯᕋᖅᐳᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐋᖅᑭᒃᓱᐃᖁᓪᓗᒋᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᕌᓇᔭᐅᔪᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᓇᓄᖏᓐᓄᑦ. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐃᒪᐃᖁᔨᕗᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᕌᓇᔭᐅᔪᑦ **ᒫᓐᓇ ᐅᓄᕐᓂᖏᑦ ᑲᔪᓯᑎᓪᓗᒋᑦ**.

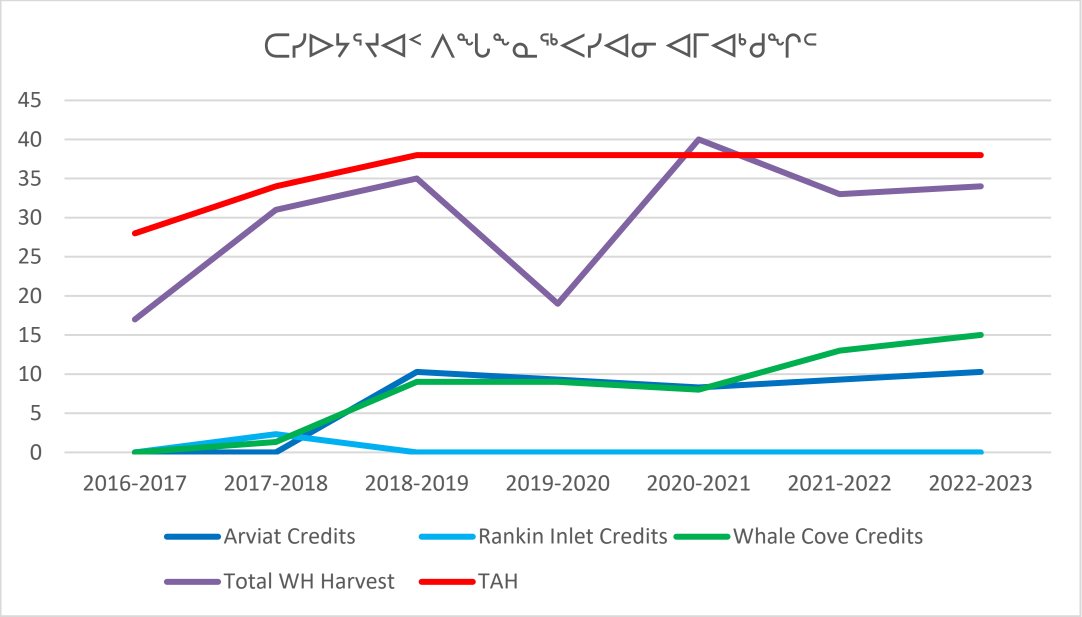
ᐃᓚᒋᐊᕈᑎᖓ 1



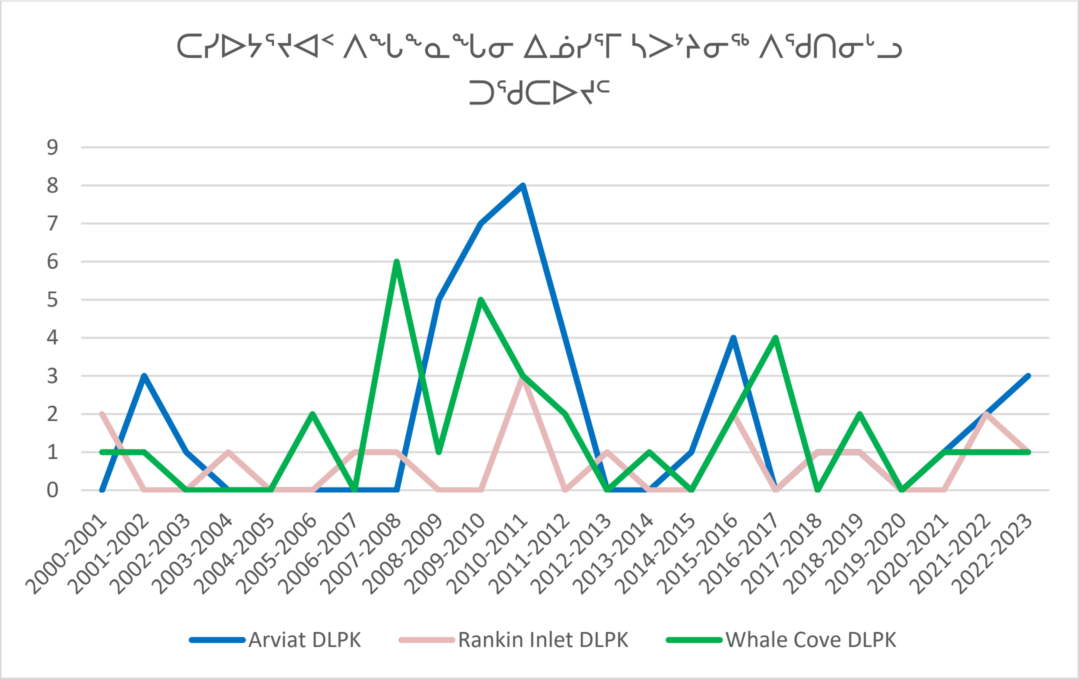
ᐊᔾᔨᙳᐊᖅ 1. ᖃᓄᐃᓕᖓᓂᖏᑦ ᓄᓇᕗᒻᒥ ᓇᓄᐃᑦ (WH = ᑕᓯᐅᔭᖅᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ, SH = ᑕᓯᐅᔭᖅᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ).



ᐊᔾᔨᙳᐊᖅ 2. ᖃᓄᐃᓕᖓᓂᖏᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖓᓂ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH), ᐊᑐᖅᑕᐅᓪᓚᕆᒃᑐᑦ ᐊᑐᖅᑕᐅᒐᔪᒃᑐᓪᓗ ᐊᖑᔭᐅᔪᓐᓇᖅᑐᑦ 2000–ᒥ ᐱᒋᐊᖅᓱᓂ.



ᐊᔾᔨᙳᐊᖅ 3. ᖃᓄᐃᓕᖓᓂᖏᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᖕᓇᖅᐸᓯᐊᓂ ᓇᓐᓄᓕᒃᓴᐃᑦ ᐊᒥᐊᒃᑯᖏᑦ 2016–ᒥ ᐱᒋᐊᖅᓱᓂ.



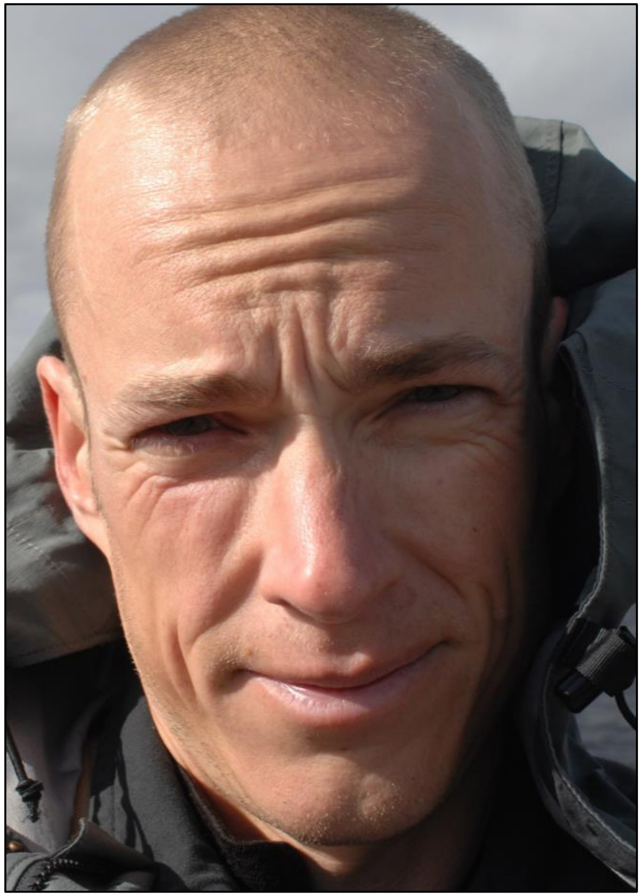
ᐊᔾᔨᙳᐊᖅ 4. ᖃᔅᓯᐅᓂᖏᑦ ᐃᓅᓯᕐᒥᑦ ᓴᐳᔾᔨᓂᕐᒧᑦ ᐱᕐᑯᑎᓂᒡᒍ ᓇᓄᐃᑦ ᑐᕐᑯᑕᐅᔪᑦ ᑕᓯᐅᔮᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ 2000–ᒥ ᐱᒋᐊᖅᓱᓂ.

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ
Department of Environment
Avatiliqiyikkut
Ministère de l’Environnement

ᓄᓇᓕᖕᓂᑦ ᑐᓴᕋᓱᖕᓃᑦ ᑕᓯᐅᔭᖅᔪᐊᑉ ᐱᖓᖕᓇᖓᓂᑦ ᓇᓄᖅᓂᒃ ᖃᖓᑕᓲᕋᑖᒃᑯᑦ ᓈᓴᐃᓂᖅᒧᑦ 2021

ᐱᒋᐊᕈᑎᒃᓴᑦ

ᐊᓖᓴ ᐳᕼᐋᖅᑦ
ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑎ
ᐊᕙᑎᓕᕆᔨᒃᑯᑦ
ᓄᓇᕗᑦ ᒐᕙᒪᖓ

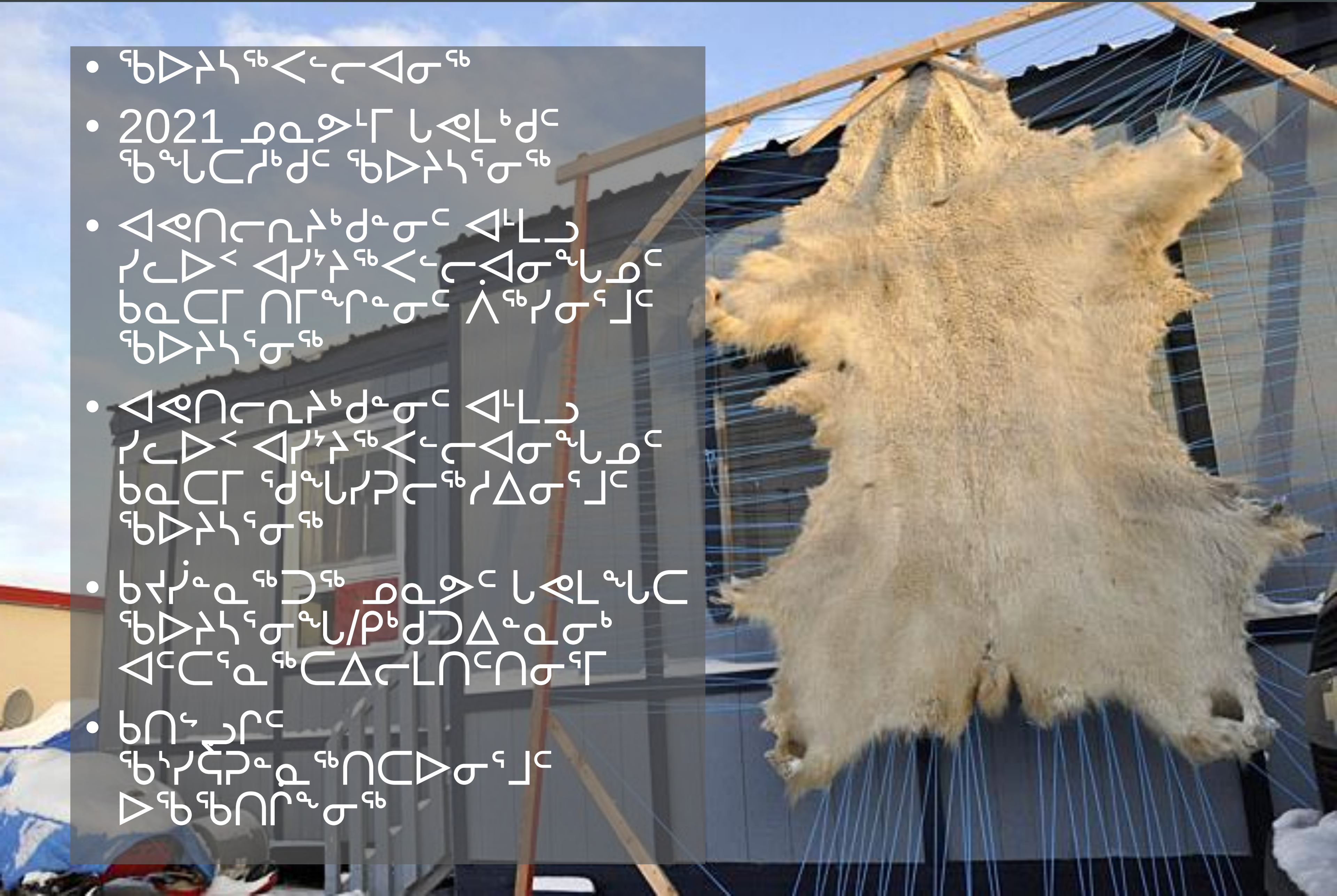


ᐃᕖᓐ ᓇᑦᓱᑦᓴᓐ
ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑎ
ᐊᕙᑎᓕᕆᓂᕐᒧᑦ ᐊᒻᒪ ᓯᓚᐅᑉ
ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥᑦ

ᐊᓯᐊᓄᐊᕐᓂᖅ

- ᑕᐃᒪᙵᑦ 2021-ᒥᑦ ᐊᓯᐊᙳᖅᓯᒪᔪᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓂᑦ
  - ᐃᓅᔪᓐᓃᒃᑳᓚᓚᐅᖅᑐᖅ ᓇᓄᕐᓄᑦ ᐆᒪᔪᓕᕆᔨ ᒫᑲᔅ ᑕᐃᒃ ᐱᓂᕐᖢᐊᕿᓂᕐᒥᑦ
  - ᒪᕐᕉᒃ ᓇᓄᕐᓄᑦ ᐆᒪᔪᓕᕆᔩᑦ ᓄᖅᖃᓚᐅᖅᑐᑦ ᐊᕐᕌᓂ
  - ᒪᕐᕉᒃ ᓄᑖᒃ ᓇᓄᕐᓄᑦ ᐆᒪᔪᓕᕆᔩᑦ ᑲᔪᓯᔪᑦ ᑐᕌᖓᔪᒥᑦ

ᖃᓄᐃᓕᖓᓂᖓ



- ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ
- 2021 ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᖃᖓᑕᓱᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ
- ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᓂᖅ
- ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᖁᖓᓯᕈᓕᖅᓱᐃᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᓂᖅ
- ᑲᔪᓰᓐᓇᖅᑐᖅ ᓄᓇᕗᑦ ᒐᕙᒪᖓᑕ ᖃᐅᔨᓴᕐᓂᖓ/ᑭᒃᑯᑐᐃᓐᓇᓂᒃ ᐊᑦᑕᕐᓇᖅᑕᐃᓕᒪᑎᑦᑎᓂᕐᒥ
- ᑲᑎᔅᓗᒋᑦ ᖃᔅᓯᐋᕈᓐᓇᖅᑎᑕᐅᓂᕐᒧᑦ ᐅᖃᖃᑎᒌᖕᓂᖅ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1. ᓴᖅᑭᑦᑎᑐᑎᒃ ᖃᐅᔨᓴᕐᕕᒃᓴᒥᒃ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1.

2. ᐅᖃᖃᑎᖃᕐᓗᑎᒃ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐋᖅᑭᒃᓯᒪᖕᒐᓂᒃ ᑐᓴᕐᓗᑎᒡᓗ ᐅᖃᐅᓯᐅᔪᓂᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1.

3. ᐃᓚᓕᐅᑎᓪᓗᒋᑦ ᑭᐅᔾᔪᑕᐅᔪᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐋᖅᑭᒃᓯᒪᓂᕆᔭᐅᔪᒧᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1.

4. ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓗᑎᑦ

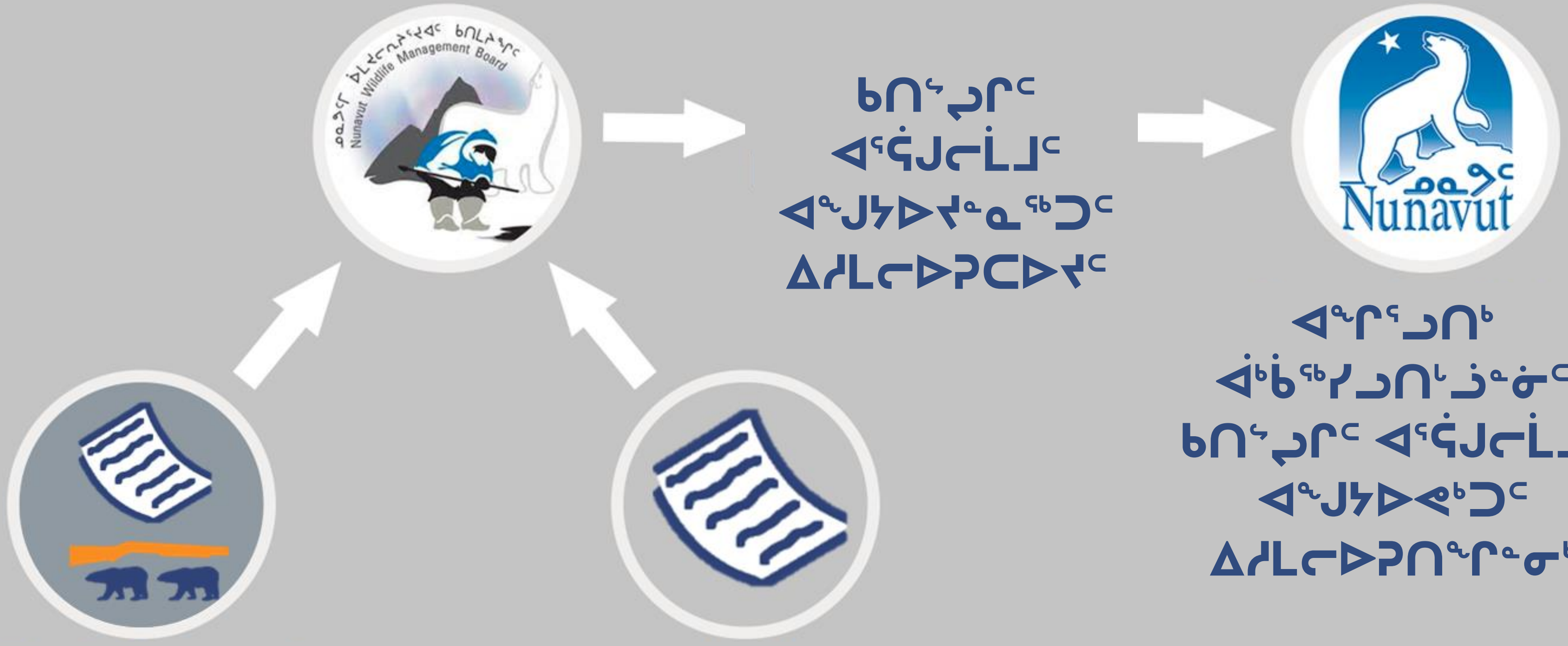
ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

5. ᕿᒥᕐᕈᓂᖅ ᐊᒻᒪ ᑎᑎᕋᕐᓂᖅ ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᒥᒃ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

6. ᑐᓴᕋᓱᖕᓂᖅ ᑭᖑᓪᓕᖅᐹᒥᑦ ᐅᓂᒃᑳᒥᑦ

ᖃᐅᔨᓴᖅᐸᓪᓚᐊᓂᖅ



ᒥᓂᔅᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ

ᑲᑎᖦᖢᒋᑦ
ᐊᕐᕌᒍᓕᒫᒧᑦ
ᐊᖑᔾᔭᐅᔪᓐᓇᖅᑐᑦ
ᐃᓱᒪᓕᐅᕈᑕᐅᔪᑦ

ᐊᖏᕐᓱᑎᒃ
ᐋᒃᑳᖅᓯᓱᑎᒡᓘᓐᓃᑦ
ᑲᑎᖦᖢᒋᑦ ᐊᕐᕌᒍᓕᒫᒧᑦ
ᐊᖑᔾᔭᐅᕙᒃᑐᑦ
ᐃᓱᒪᓕᐅᕈᑎᖏᓐᓂᒃ

ᑭᖑᓪᓕᖅᐹᖅ ᐅᓂᒃᑳᖅ ᐊᒻᒪ
ᑲᑎᓪᓗᒋᑦ ᐊᕐᕌᒍᓕᒫᖅᓯᐅᑎᑦ
ᖃᔅᓯᐅᓂᖏᑦ
ᐊᑐᓕᖅᑯᔭᐅᔪᑦ

ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ
ᑭᖑᓪᓕᖅᐹᖅ ᐅᓂᒃᑳᖅ

7. ᑲᑎᖦᖢᒋᑦ ᐊᖑᔾᔭᐅᔪᓐᓇᖅᑐᑦ ᐃᓱᒪᓕᐅᕈᑎᖏᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

ᒥᓂᔅᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᖓᓄᑦ

ᐊᖏᖅᑕᐅᔪᑦ

ᐊᖏᕐᓗᑎᒃ ᐋᒃᑳᕐᓗᑎᒡᓘᓐᓃᑦ
ᑲᑎᖦᖢᒋᑦ ᐊᕐᖀᒍᓕᒫᒍᑦ
ᐊᖑᔭᐅᕙᒃᑐᑦ
ᐃᓱᒪᓕᐅᕈᑎᖏᓐᓂᒃ

ᐊᕕᒃᑐᖅᓯᒪᔪᓂ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔩᖏᑦ

ᐋᖅᑭᒃᓯᓗᑎᑦ ᖃᓯᓂᒃ ᓂᕚᖓᑖᓂᒃ
ᑕᒪᕐᒥᒃ ᓄᓇᓖᑦ ᐱᓂᐊᕐᒪᖔᑕ

ᓄᓇᓕᖕᓄᑦ ᓂᕚᙵᑖᑦ
ᓇᒃᓯᐅᔾᔭᐅᔪᑦ

7. ᑲᑎᖦᖢᒋᑦ ᐊᖑᔭᐅᔪᓐᓇᖅᑐᑦ ᐃᓱᒪᓕᐅᕈᑎᖏᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

7.

ᒥᓂᔅᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ

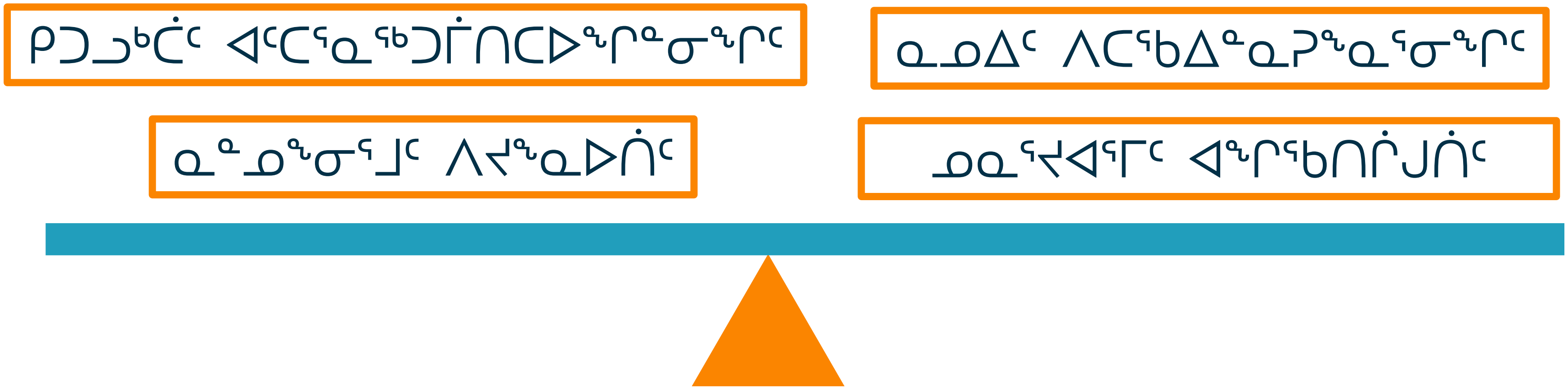
ᐊᖏᖅᑕᐅᔪᑦ

ᐊᕕᒃᑐᖅᓯᒪᔪᓂ ᐆᒪᔪᓕᕆᔨᕐᑯᑦ ᑲᑎᒪᔨᖏᑦ

ᓇᓄᒃᑕᐅᔪᓂᒃ ᐊᐅᓚᑦᑎᓂᖅ

ᐱᓕᕆᐊᖓᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ

- ᐃᒡᓗᕖᖓᓕᕆᓂᖅ



ᑭᑐᓗᒃᑖᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᑎᑕᐅᖏᓐᓂᖏᑦ

ᓇᓄᐃᑦ ᐱᑕᖃᐃᓐᓇᕈᓐᓇᕐᓂᖏᑦ

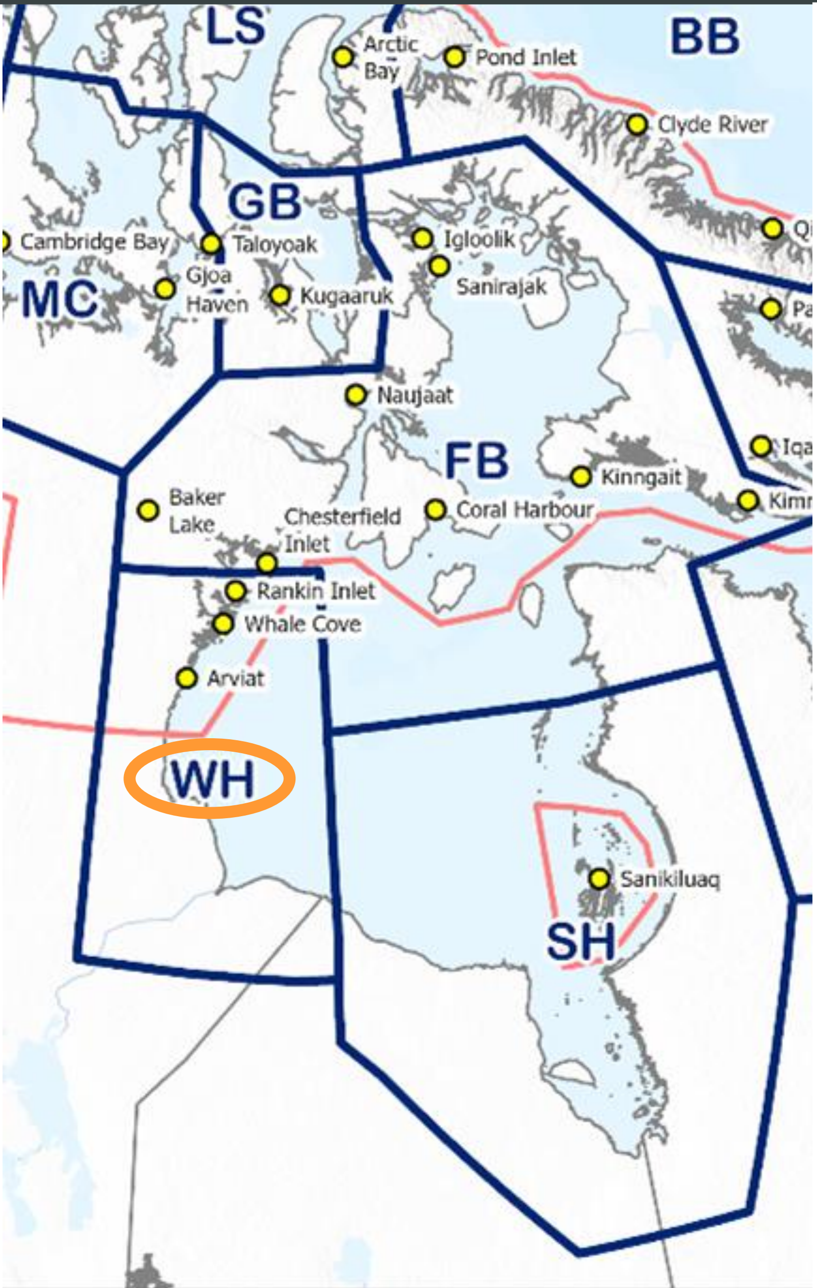
ᓇᓐᓄᕐᓂᕐᒧᑦ ᐱᔪᓐᓇᐅᑏᑦ

ᓄᓇᕐᔪᐊᕐᒥᑦ ᐊᖏᖃᑎᒌᒍᑏᑦ

ᑲᑎᓗᒃᑖᖅᓱᒥᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH)

- ᒫᓐᓇᐅᔪᖅ ᑲᑎᓗᒃᑖᖅᓱᒥᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ 38 ᓇᓄᐃᑦ ᐅᑭᐅᑕᒫᕐ.
- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᒫᓐᓇ ᐊᑐᕐᑯᔨᖕᒪᑕ **ᑲᑎᕐᓱᒥᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ 38–ᓂᑦ.**

ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᒃᑲᓐᓂᖕᒋᑦ−ᐅᓄᕐᓂᖕᒋᑦ



- ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓄᑦ 2011−ᒥ ᐊᒻᒪ 2016−ᒥ

- ᖃᐅᔨᓴᖅᑕᐅᖕᒋᓐᓇᖅᑐᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᐊᒻᒪ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖕᒐᓄᑦ ᑲᓇᑕᒥ ᑕᐃᒪᖖᒐᑦ 1980−ᖕᒋᓐᓂ
  - ᒫᓂᑑᐸᒥ ᐱᓕᕆᐊᖕᒍᓚᐅᖅᑐᑦ

- ᐊᖕᒋᖃᑎᒌᒍᑎ ᐅᓄᕐᓂᖕᒋᑦ **ᐃᓄᐃᑦ−ᓇᓄᐃᑦ ᐊᐳᖃᑦᑕᐅᑎᕙᒃᑐᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᖕᒋᓐᓇᕐᓂᖕᒋᓐᓂᒃ ᖃᖕᒪᖖᒍᖅᐸᓪᓕᐊᑎᓪᓗᒍ**

- ᖃᐅᔨᓴᖅᑕᐅᒐᔪᓪᓗᓂ ᖃᐅᔨᓂᐊᕐᓗᓂ **ᐊᓯᔾᔨᑳᓪᓚᖕᓂᐅᔪᓂᑦ** ᓇᓄᐃᑦ ᐅᓄᕐᓂᖕᒋᓐᓂᑦ

2021 AERIAL SURVEY OF THE WESTERN HUDSON BAY POLAR BEAR SUBPOPULATION

FINAL REPORT

November 16, 2022

Stephen N. Atkinson[1], John Boulanger[2], Mitch Campbell[3], Vicki Trim[4], Jasmine Ware[5]
and Amélie Roberto-Charron[6]

Submitted to meet requirements of:
Wildlife Research Permit WL 2021-0061, Wapusk National Park Research and
Collection Permit Number WAP-2021-3990

Status Report 2022, Nunavut Department of Environment, Wildlife Research Section,
Igloolik, NU

ᖁᔭᓐᓇᒦᒃ

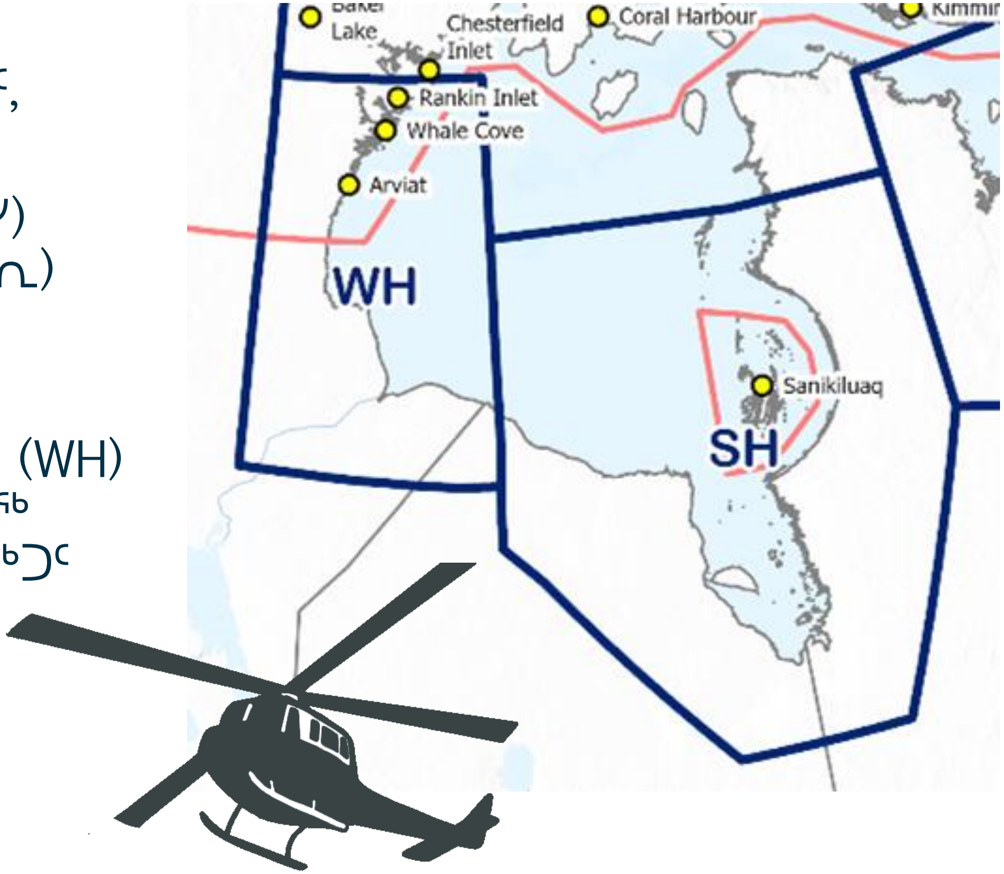
Environment and Climate Change Canada
Environnement et Changement climatique Canada

ᐱᓕᕆᐊᖅ ᐱᐊᓂᒍᖕᓇᖃᔭᓚᐅᖏᑦᑐᖅ ᐃᓚᐅᖅᑲᑕᐅᙱᑉᐸᑕ ᐊᒻᒪᓗ ᐃᑲᔪᖅᑕᐅᓚᐅᖏᑉᐸᑕ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐊᒻᒪᓗ ᐅᑯᐊ ᑕᑯᓯᒪᔪᑦ:

- ᔾᔨᖁᐅ ᒪᒃᑖᖅ
- ᑎᒧᑎ ᑯᑭᔭᒃ
- ᑎᒧᑎ ᐱᓱᒃ
- ᑭᐊᕕᓐ ᓴᐸᒍᑦ
- ᑳᓄ ᑳᒻᐹᓪ
- ᑭᓵᑕᓐ ᑕᖅᑕᖅ
- ᔫᒃ ᐹᒃᓴᑑᓐ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

- ᓈᓴᐃᓚᐅᖅᑐᑦ 2011-ᒥᑦ, 2016-ᒥᑦ, ᐊᒻᒪᓗ 2021-ᒥᑦ
- ᓈᓴᐃᓚᐅᖅᑐᑦ ᐊᑯᓪᓕᕈᕐᕕᒃ (ᐋᑎᓯ) ᓄᙳᐊᓂᑦ – ᐊᒥᕐᐃᔭᕐᕕᒃ (ᓯᑎᐱᓇ) ᐱᒋᐊᓵᕐᓂᖕᒐᓄᑦ
    - ᓇᓄᐃᑦ ᓄᓇᒦᑦᑐᑦ
- ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ (WH) ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ (SH) ᐊᑕᐅᑦᑎᒃᑯᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ



ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

- ᑲᑎᖅᓱᕐᓗᒋᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᑦ **ᒥᒃᓴᐅᓴᖕᓂᐊᕐᓗᒋᑦ ᐊᒥᓲᓂᖏᑦ** ᐆᒪᔪᑦ
- **ᐅᖓᓯᒋᔭᒥ ᖃᐅᔨᓴᕐᓂᖅ** ᐊᒻᒪ **ᒪᕐᕉᖕᓂᒃ ᖁᙱᐊᖅᑎᖃᕐᓗᑎᒃ** ᖃᐅᔨᓴᓛᖅᑐᑦ
- **ᐊᒡᒍᖅᓯᒪᓂᖏᑦ ᐊᒻᒪ ᑭᑉᐹᓇᒃᓯᓕᖅᓯᒪᔪᑦ** ᑐᙵᕕᒋᓪᓗᒋᑦ ᑭᖑᓂᖔᑦᑎᓐᓂ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᓚᐅᖅᑐᑦ, ᐊᖑᔭᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᑦ, ᐃᓚᓕᐅᑎᓂᐊᖅᐸᕗᓪᓗ ᓄᓇᓕᖕᓂ ᑭᐅᔾᔪᑕᐅᖃᑦᑕᓚᐅᖅᑐᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

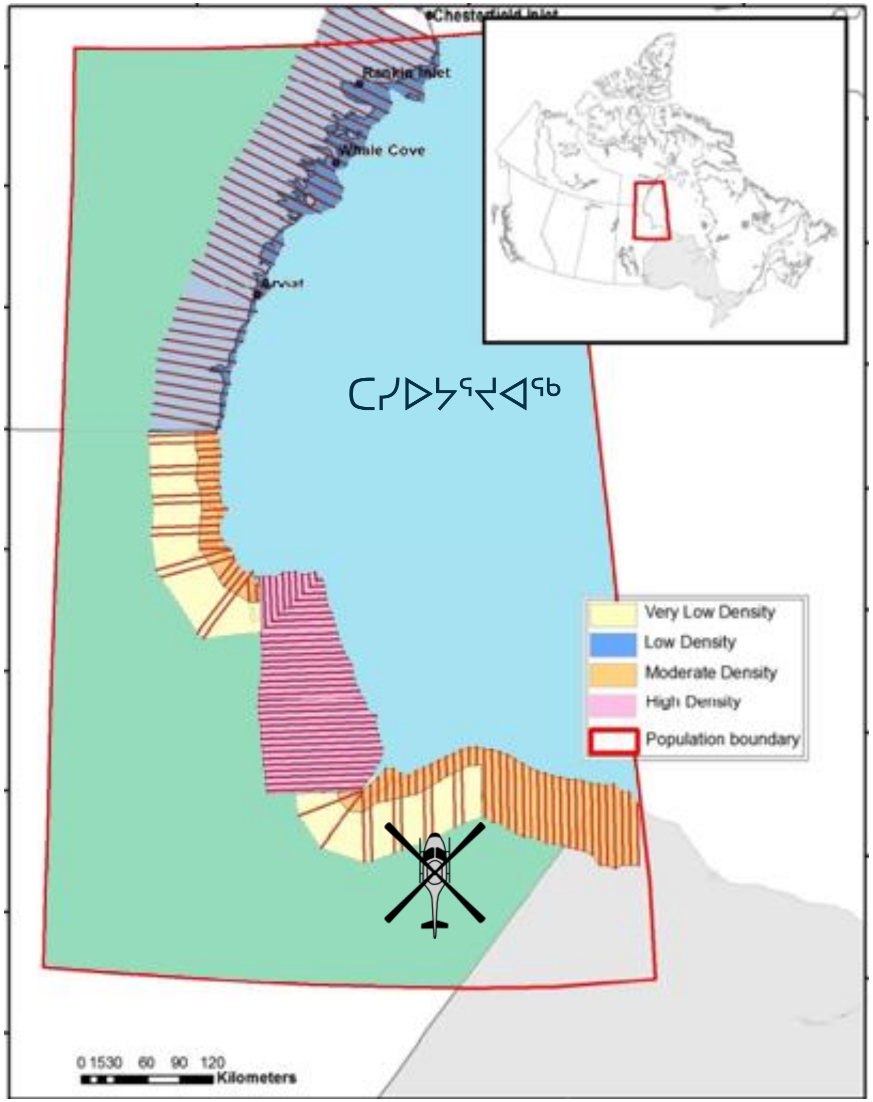
- ᐊᑐᓚᐅᖅᑐᑦ ᐊᔾᔨᖏᓐᓂᒃ ᑲᑎᓐᓂᐅᔪᓂᒃ 2016-ᒥᑦ ᖃᐅᔨᓴᐃᓂᐅᓚᐅᖅᑐᒧᑦ
- ᑲᑎᓐᓂᕐᓂᑦ 2016-ᒥᑦ ᖃᐅᔨᓚᐅᖅᑐᑦ ᐊᑐᖅᖢᑎᒃ:
  - 2011-ᒥᑦ ᓈᓴᐃᓂᖅ ᐊᒻᒪᓗ ᖃᓄᐃᙱᓂᐅᔪᑦ
  - ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ (IQ)
  - ᐅᙱᓯᒃᑎᒋᓂᕐᒧᑦ ᐊᑐᖅᖢᑎᒃ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ (ᓯᓚᑦᑐᖅᓴᕐᕕᒡᔪᐊᖅ ᐃᐊᕀᑕ ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔾᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐅᔪᓕᕆᔾᒃᑯᑦ)
- ᑲᑎᓐᓃᑦ ᐊᕕᒃᑕᐅᔪᑦ ᓯᑕᒪᐅᓕᖅᑯᒥᑦ ᐊᒥᓲᑎᒋᓂᕐᓄᑦ:

🟨 - **ᐊᑦᑎᒃᑐᒻᒪᕆᐊᓗᒃ:** ᓄᓇᖅᐸᓯᐅᑉ ᓯᓚᑖᓂᑦ ᐅᐊᐳᖅ ᒥᕐᒍᐃᖅᓯᕐᕕᖕᒥᑦ

🟦 - **ᐊᑦᑎᒃᑐᖅ:** ᐅᐊᖕᓇᖅᐸᓯᙵᓂᑦ ᐱᖓᖕᓇᖅᐸᓯᙵᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ

🟧 - **ᕿᑎᐊᓃᑦᑐᖅ:** ᐅᐊᖕᓇᖕᒐᓂᑦ ᐊᒻᒪᓗ ᐱᖓᖕᓇᖕᒐᓂᑦ ᑰᒡᔪᐊᑉ ᑯᖕᒐᓂᑦ

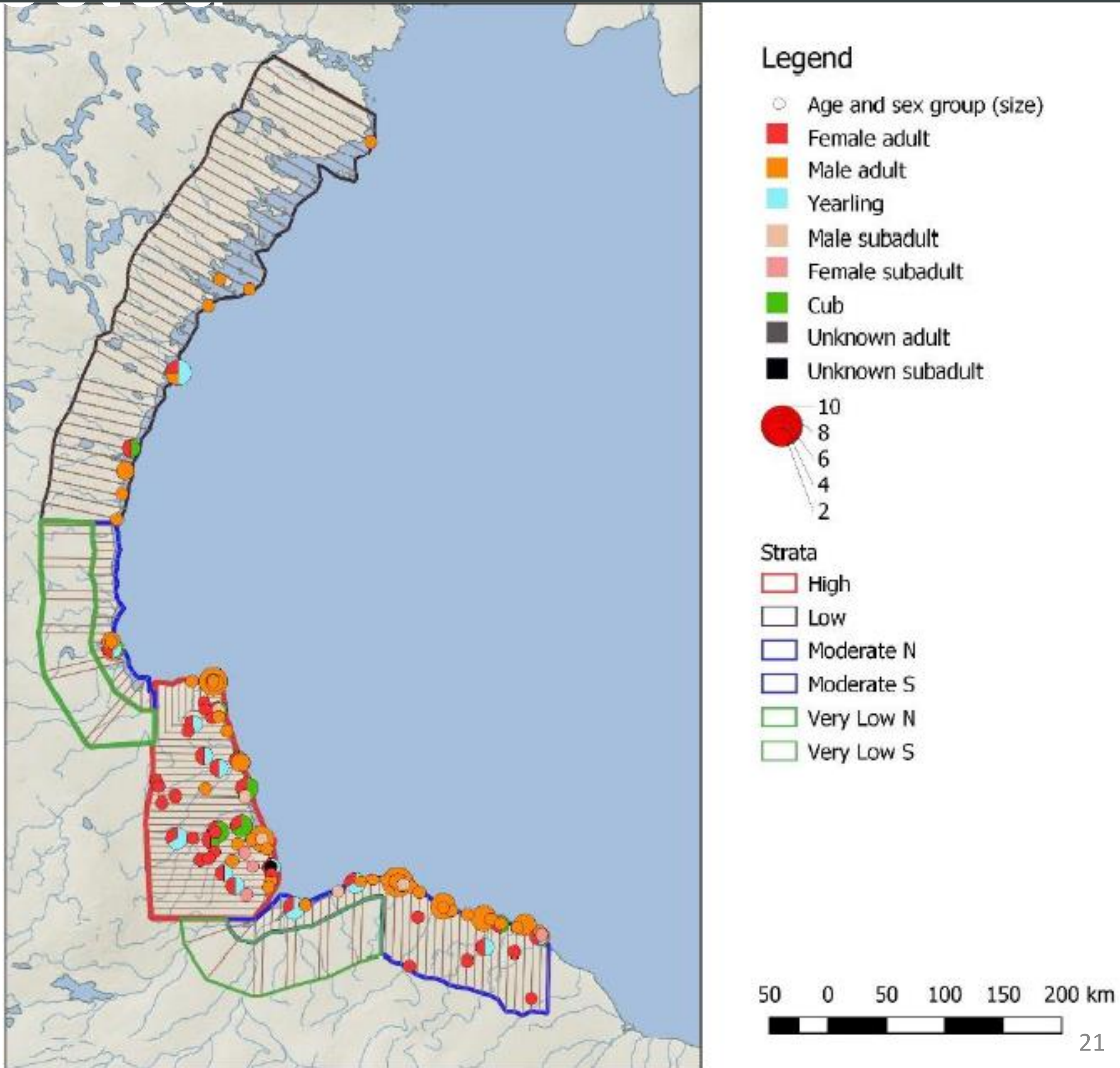
🟪 - **ᖁᑦᑎᒃᑐᖅ:** ᐅᐊᐳᖅ ᒥᕐᒍᐃᖅᓯᕐᕕᖕᒥᑦ, ᓯᒡᔭᒥᑦ ᐊᑯᓐᓂᖕᒐᓂᑦ ᑰᒡᔪᐊᑉ ᐊᒻᒪᓗ ᓂᐅᑦᓴᓐ ᑰᑉ



ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᒃᑲᓐᓂᖏᑦ–ᐅᓄᕐᓂᖏᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

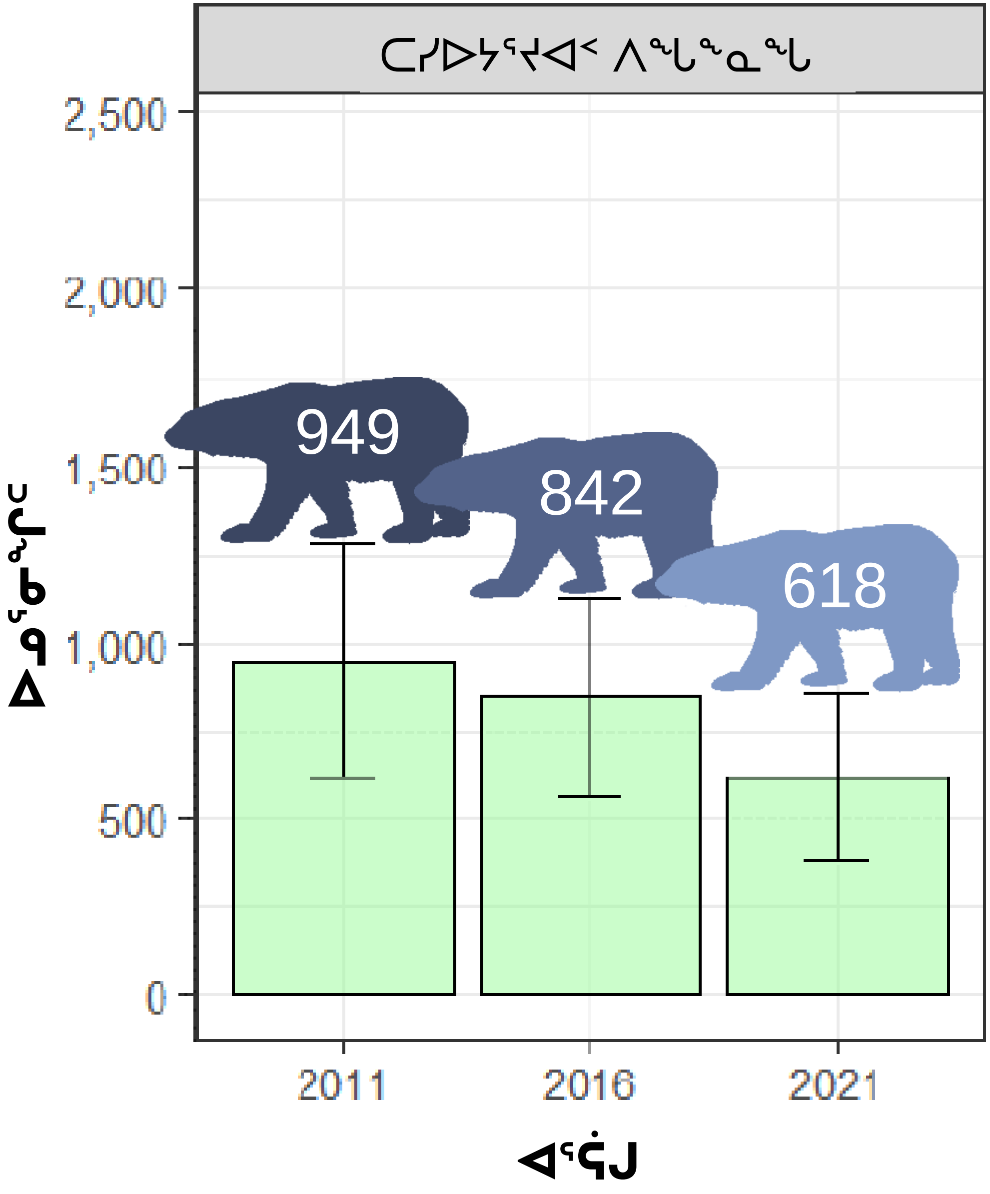
- 17,000 ᑭᓛᒥᑕᓂᒃ ᖃᖓᑕᓂᖅ
- ᑎᑭᐅᑎᔪᑦ ᓇᓄᕐᓄᑦ 194, 125-ᖑᔪᓂᑦ ᑲᑎᕐᒪᔪᓂᑦ ᖃᐅᔨᔭᐅᔪᓂᑦ
- ᓇᓚᐅᑦᑖᖅᑕᐅᔪᖅ: 618 ᓇᓄᐃᑦ
- 8%-ᖏᑦ ᓇᓄᐃᑦ ᓄᓇᕗᒻᒥᑦ
- ᐊᖑᓴᓪᓗᐃᑦ ᓯᒡᔭᖅᐸᓯᖕᒦᒐᔪᓛᖑᔪᑦ
- ᐊᕐᓇᓪᓗᐃᑦ ᓄᓇᖅᐸᓯᖕᒦᒐᔪᓛᖑᔪᑦ



ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᒃᑲᓐᓂᖏᑦ-ᐅᓄᕐᓂᖏᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

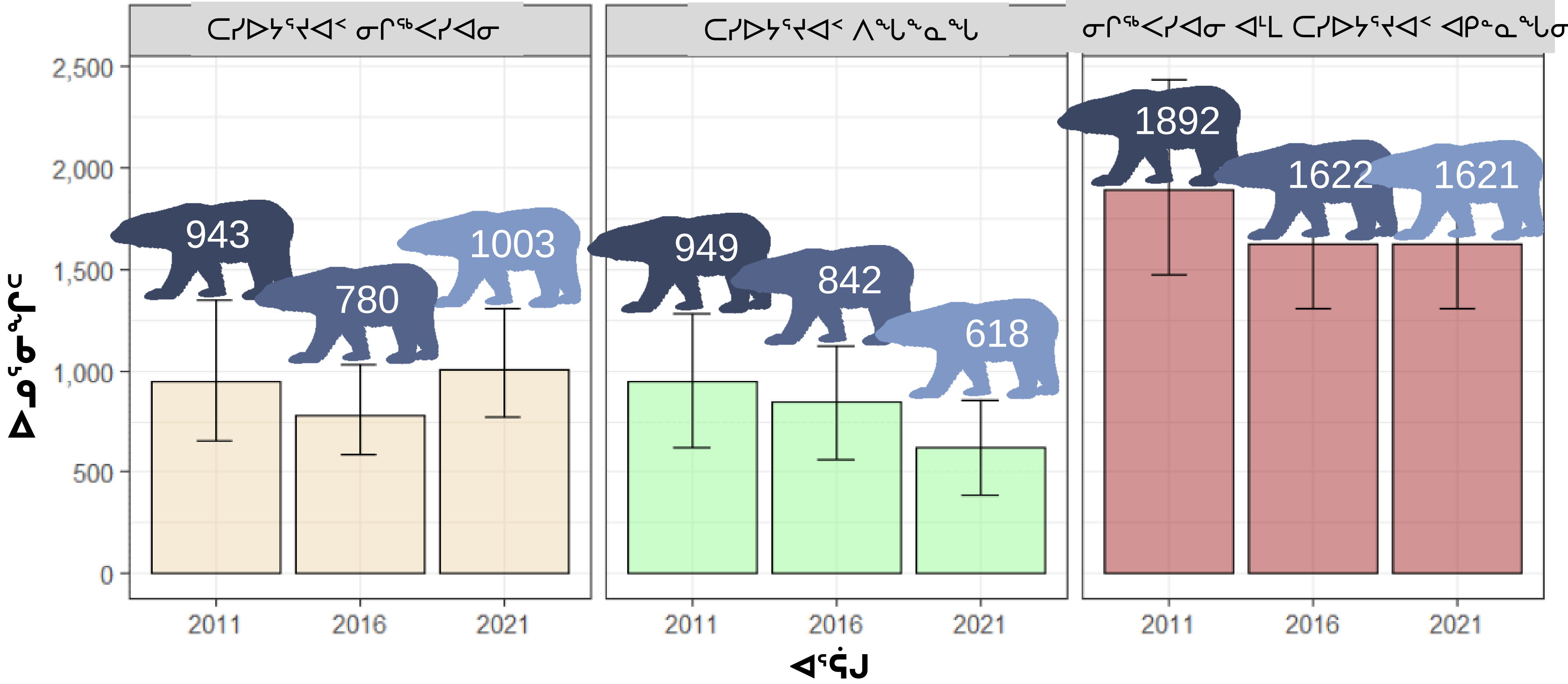
- 17,000 ᑭᓛᒥᑕᓂᒃ ᖃᖓᑕᓂᖅ
- ᑎᑭᐅᑎᔪᑦ ᓇᓄᕐᓄᑦ 194, 125-ᖑᔪᓂᑦ ᑲᑎᕐᒪᔪᓂᑦ ᖃᐅᔨᔭᐅᔪᓂᑦ
- ᓇᓚᐅᑦᑖᖅᑕᐅᔪᖅ: 618 ᓇᓄᐃᑦ
- 8%-ᖏᑦ ᓇᓄᐃᑦ ᓄᓇᕗᒻᒥᑦ
- ᐊᖑᓯᖅᑐᐃᑦ ᓯᒡᔭᖅᐸᓯᖕᒦᒐᔪᓛᖑᔪᑦ
- ᐊᕐᓇᖅᑐᐃᑦ ᓄᓇᖅᐸᓯᖕᒦᒐᔪᓛᖑᔪᑦ



ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᒃᑲᓐᓂᖏᑦ–ᐅᓄᕐᓂᖏᑦ

ᖃᖓᑕᓲᒃᑯᑦ
ᐊᒥᓲᓂᖏᓐᓂᑦ
ᖃᐅᔨᓴᕈᑎᑦ

- ᐱᔾᔪᑕᐅᔪᖅ ᐊᒥᓱᙳᓇᐊᕐᓂᕐᒧᑦ?
  - ᐅᑎᖅᑐᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ?



ᐲᖅᓯᓂᖅ ᖃᐅᔨᓴᖅᑕᐅᔪᒃᓴᒥᒃ ᓯᓂᒃᑎᑕᐅᓗᑎᒃ


Environment and Climate Change Canada
Environnement et Changement climatique Canada

Distributional shifts of polar bears (*Ursus maritimus*) in Hudson Bay in relation to sea ice dynamics, 2017-2022 Final Report

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[2] Ontario Ministry of Natural Resources and Forestry, DNA Building B217, Trent University, 2140 East Bank Drive, Peterborough, ON K9L 1Z8
[3] Manitoba Department of Natural Resources and Northern Development, Water Stewardship and Biodiversity Division, Wildlife and Fisheries Branch, Box 28, 59 Elizabeth Drive, Thompson, MB R8N 1X4
[4] Department of Biological Sciences, University of Alberta, Edmonton, AB T6G 2E9

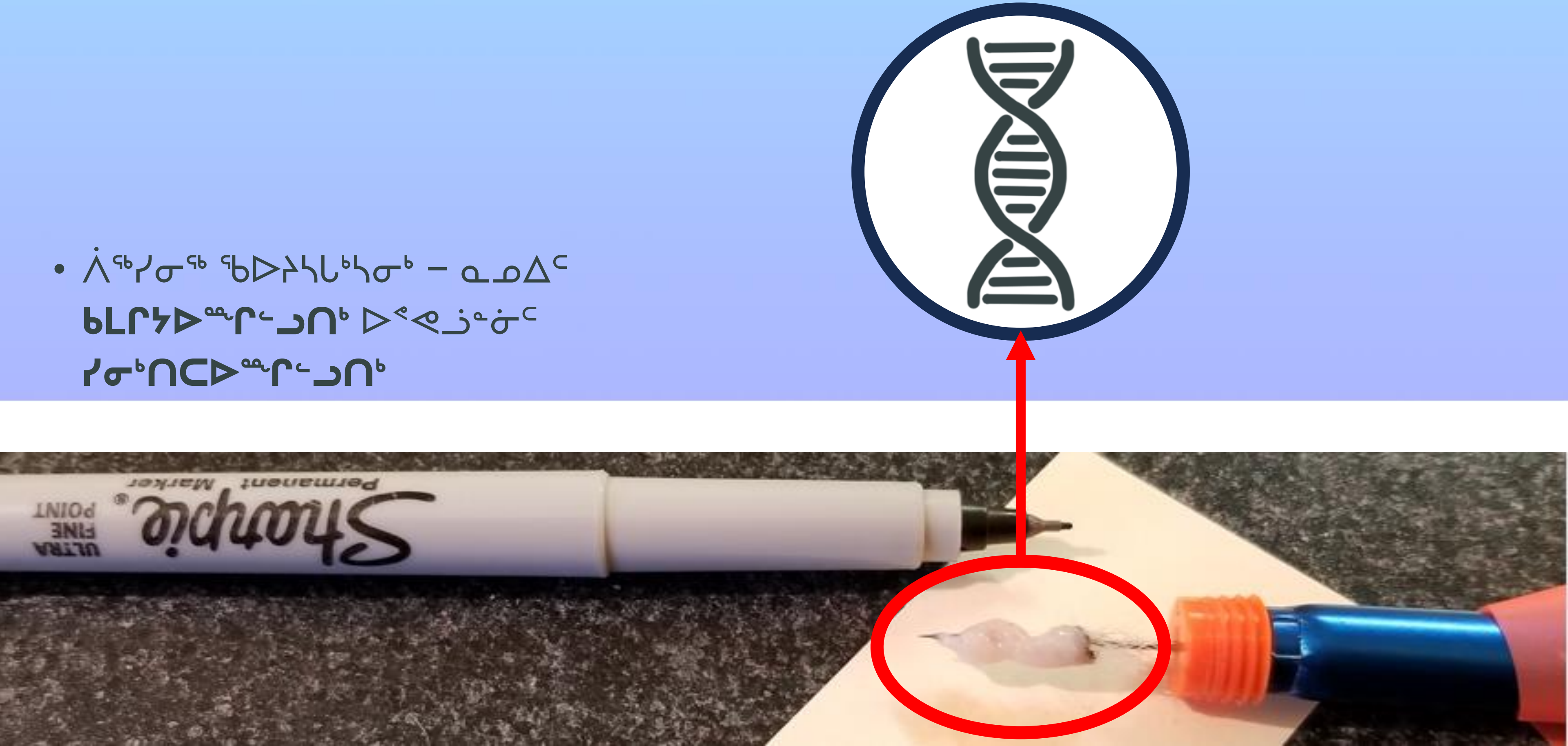
ᑕᐃᕕᑦ ᒪᒌᓯ
ᓇᓄᓕᕆᔨ
ᐊᕿᑎᓕᕆᓂᕐᒧᑦ ᐊᒻᒪ ᓯᓚᐅᑉ
ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥᑦ

ᖃᐅᔨᓴᒐᒃᓴᓂᒃ ᓯᓂᓕᖅᑎᑕᐅᙱᓪᓗᓂ

ᖃᐅᔾᓴᒐᒃᓴᓂᒃ ᓯᓂᓚᖅᑎᑕᐅᙱᕐᓗᓂ

ᓯᕘᓕᖅᓱᕈᑏᕐ/ᓇᑭᙶᕐᓂᖏᑦ

- ᐲᖅᓯᓂᖅ ᖃᐅᔾᓴᒐᒃᓴᓂᒃ – ᓇᓄᐃᑦ **ᑳᒪᒋᔭᐅᙱᕐᓗᑎᒃ** ᐅᕝᕙᓘᓐᓃᑦ **ᓯᓂᒃᑎᑕᐅᙱᕐᓗᑎᒃ**




Hudson
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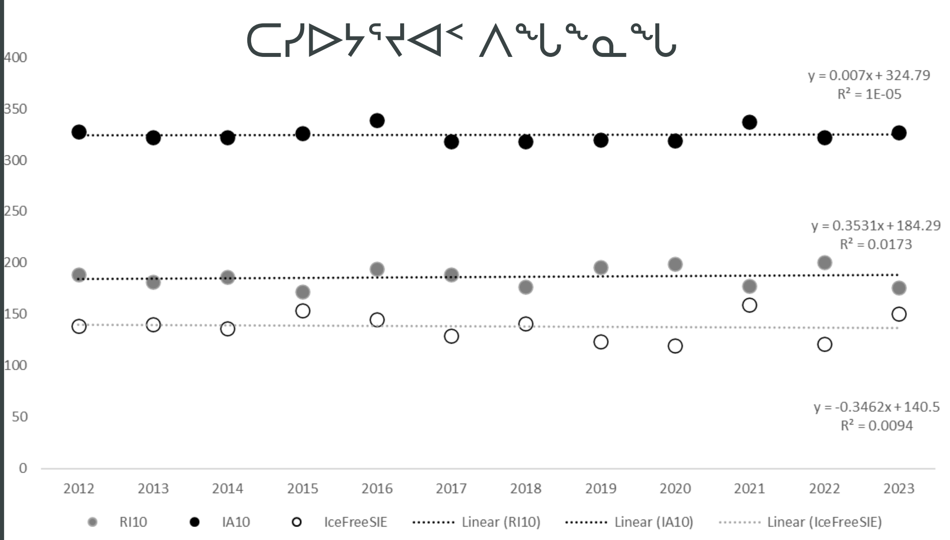
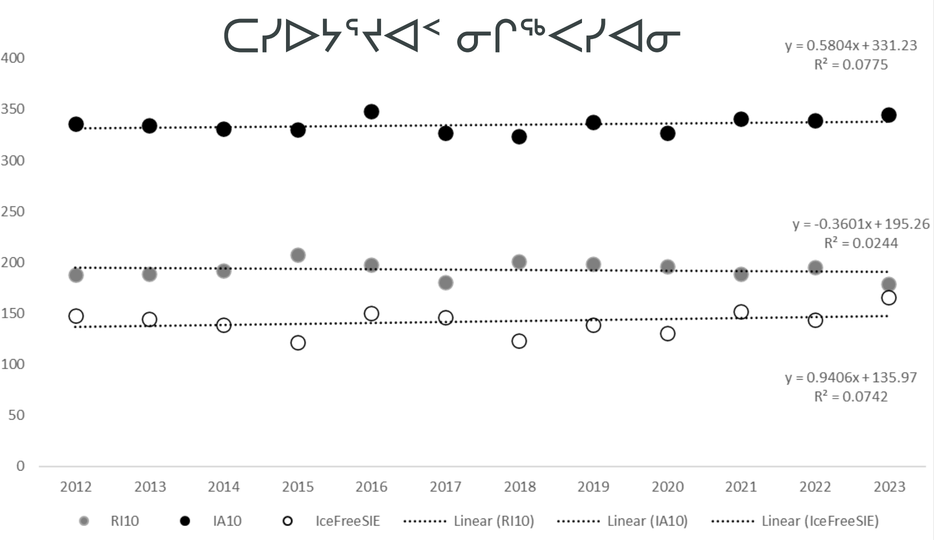
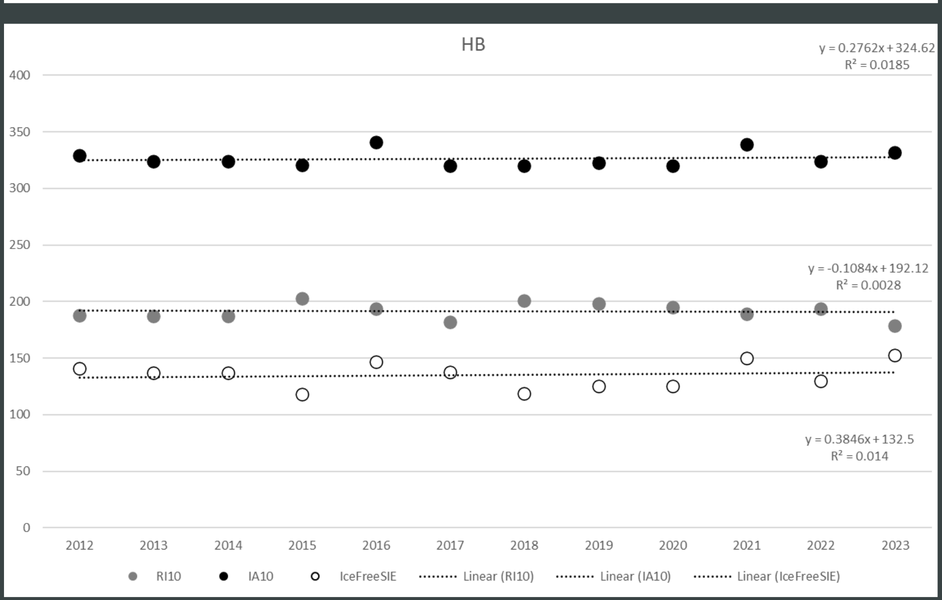
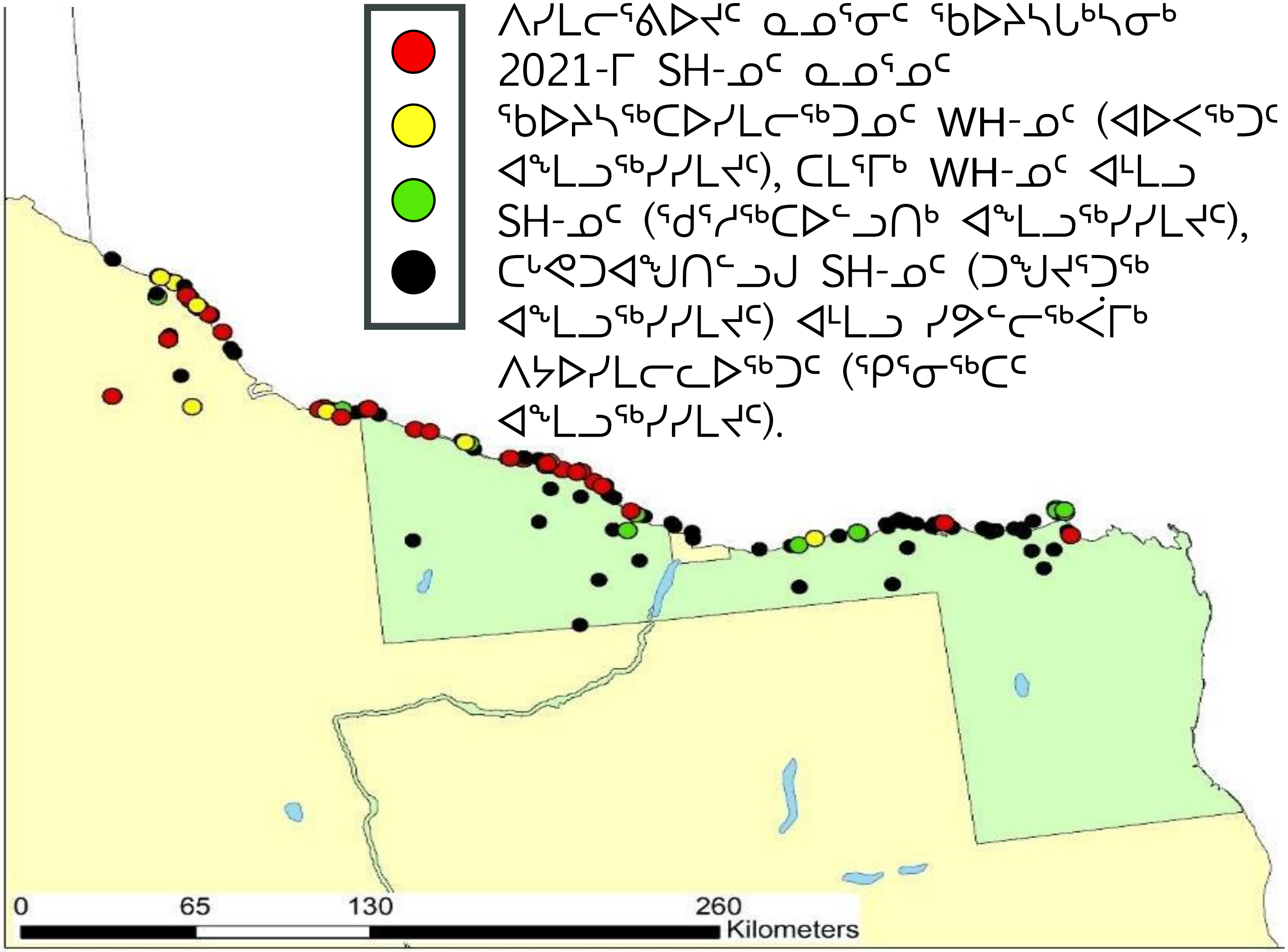
ᖃᓄᐃᓕᕐᓂᖕᒋᑦ - ᑕᕆᐅᒥ ᓱᑯᑦ ᐃᓕᖕᒐᓐᓂᕆᔭᖕᒋᑦ





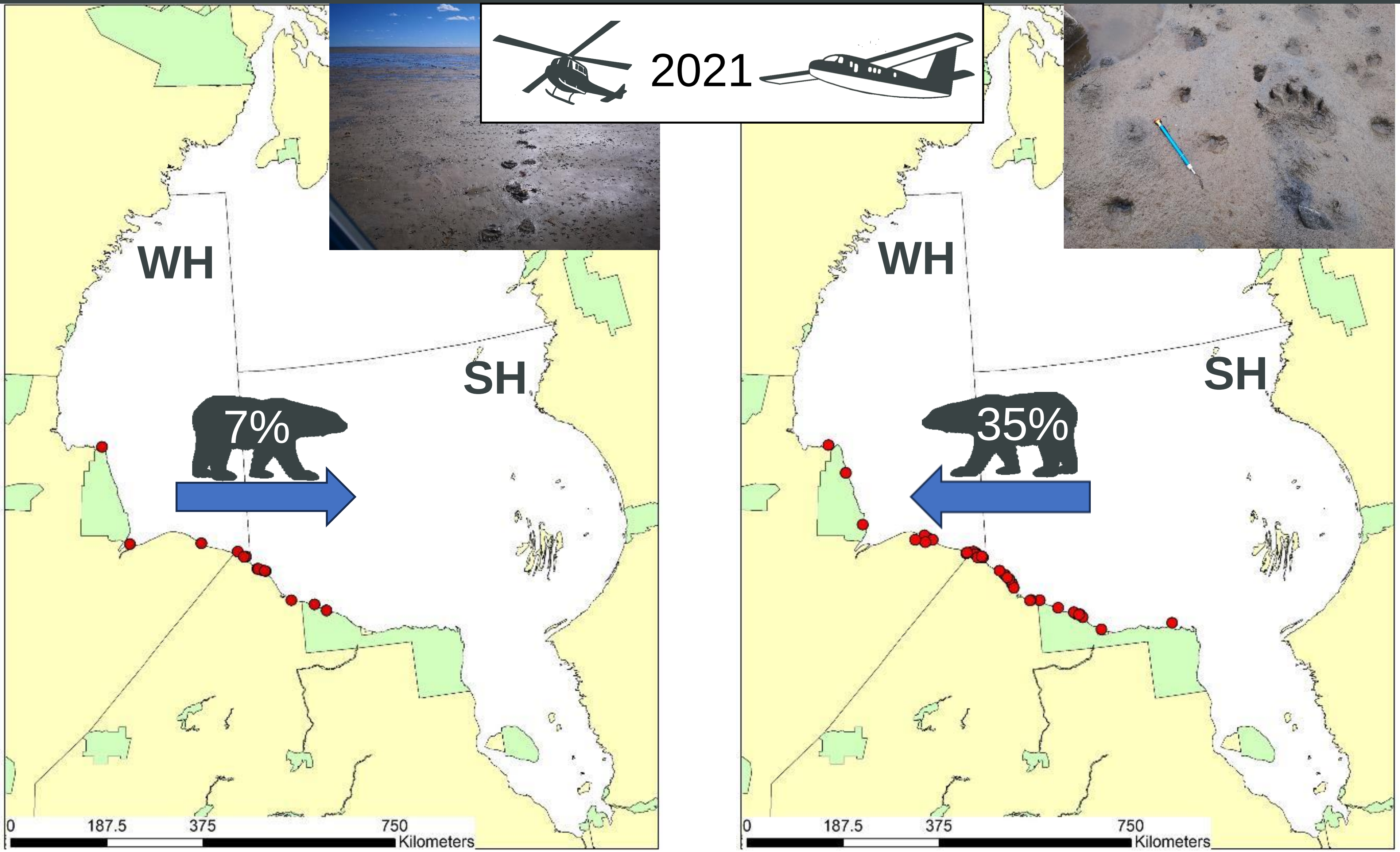


ᖃᓄᐃᓕᖅᓂᖏᑦ - 2021 SH-ᓄᑦ ᒫᓐᓇᒧᑦ ᑎᑭᓪᓗᒍ

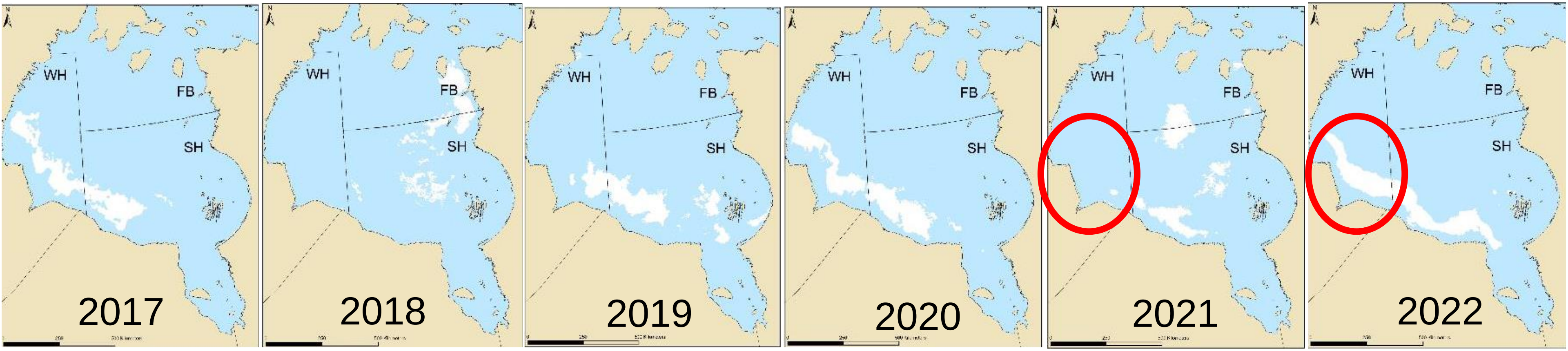


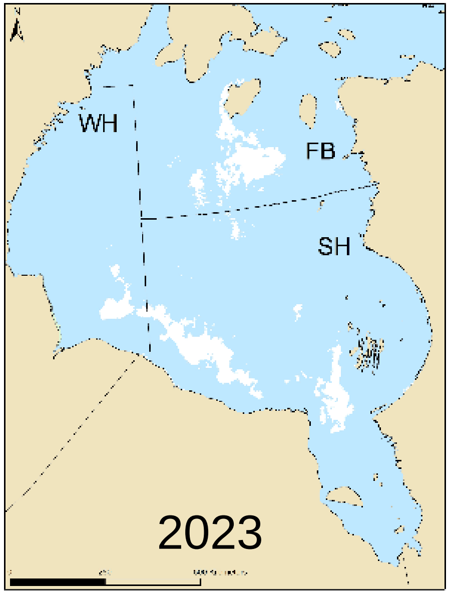
ᑎᑎᕋᐅᔭᖅᓯᒪᔪᖅ 6. ᖃᐅᔨᓴᒐᒃᓴᓂᒃ ᓇᑭᑦ ᐱᓯᒪᓕᕐᕕᐅᔪᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᒐᒃᓴᓂᒃ 2021-ᒥ SH-ᓄᑦ ᓇᓄᕐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᓕᖅᑐᓄᑦ WH-ᓄᑦ (ᐊᐅᐸᖅᑐᑦ ᐊᓐᒪᓗᖅᓯᓯᒪᔪᑦ), ᑕᒪᕐᒥᒃ WH-ᓄᑦ ᐊᒻᒪᓗ SH-ᓄᑦ (ᖁᕐᓱᖅᑕᐅᓪᓗᑎᒃ ᐊᓐᒪᓗᖅᓯᓯᒪᔪᑦ), ᑕᒡᕙᑐᐊᖑᑎᓪᓗᒍ SH-ᓄᑦ (ᑐᖑᔪᕐᑐᖅ ᐊᓐᒪᓗᖅᓯᓯᒪᔪᑦ) ᐊᒻᒪᓗ ᓯᕗᓪᓕᖅᐹᒥᒃ ᐱᔭᐅᓯᒪᓕᓚᐅᖅᑐᑦ (ᕿᕐᓂᖅᑕᑦ ᐊᓐᒪᓗᖅᓯᓯᒪᔪᑦ).

ᖃᓄᐃᓕᕐᓂᖏᑦ - WH-ᓂᑦ SH-ᓄᑦ
ᐊᒻᒪᓗ SH-ᓂᑦ WH-ᓂᑦ 2021/2022

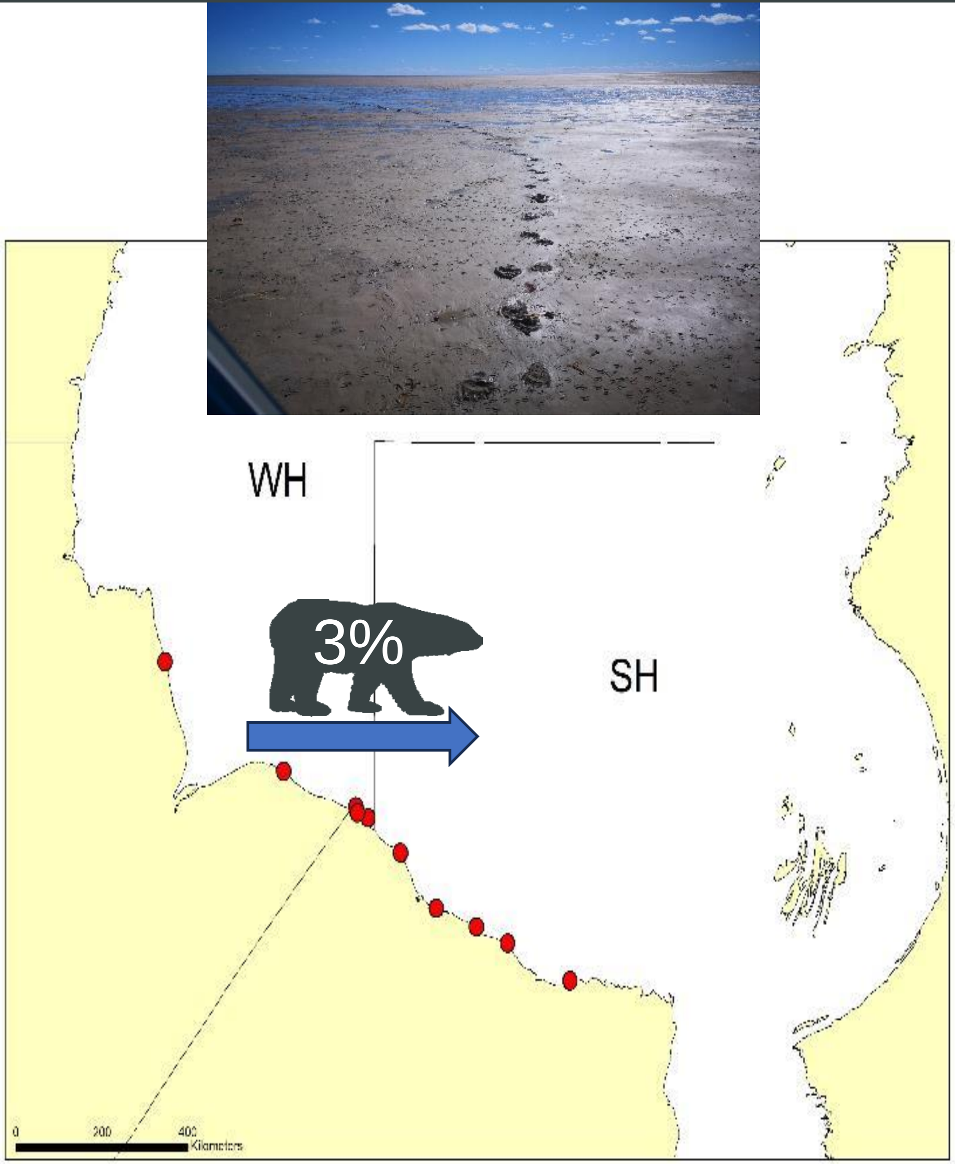


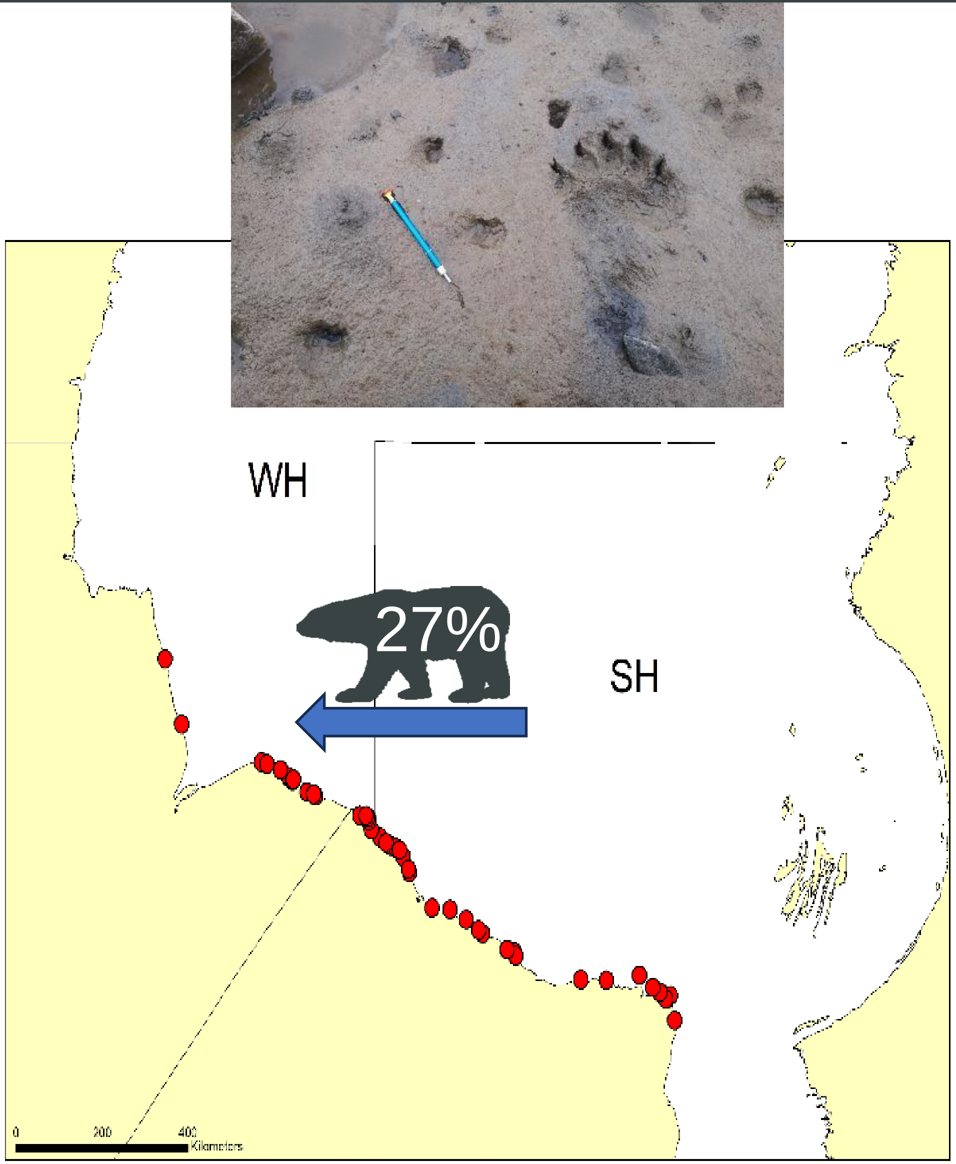
ᖃᓄᐃᓕᖅᓯᖕᒥᑦ – ᖀᖕᒍᓪᓕᖅᐸᐅᔪᖅ ᐊᕙᑎᒋᔭᐅᔪᒥᑦ ᖃᑎᓘᒃᑖᕐᓗᒥᑦ ᓯᑯᖃᕐᓂᕐᒥᑦ ᐊᐅᓯᓚᐅᖅᑳᖅᑎᓐᓇᒍ



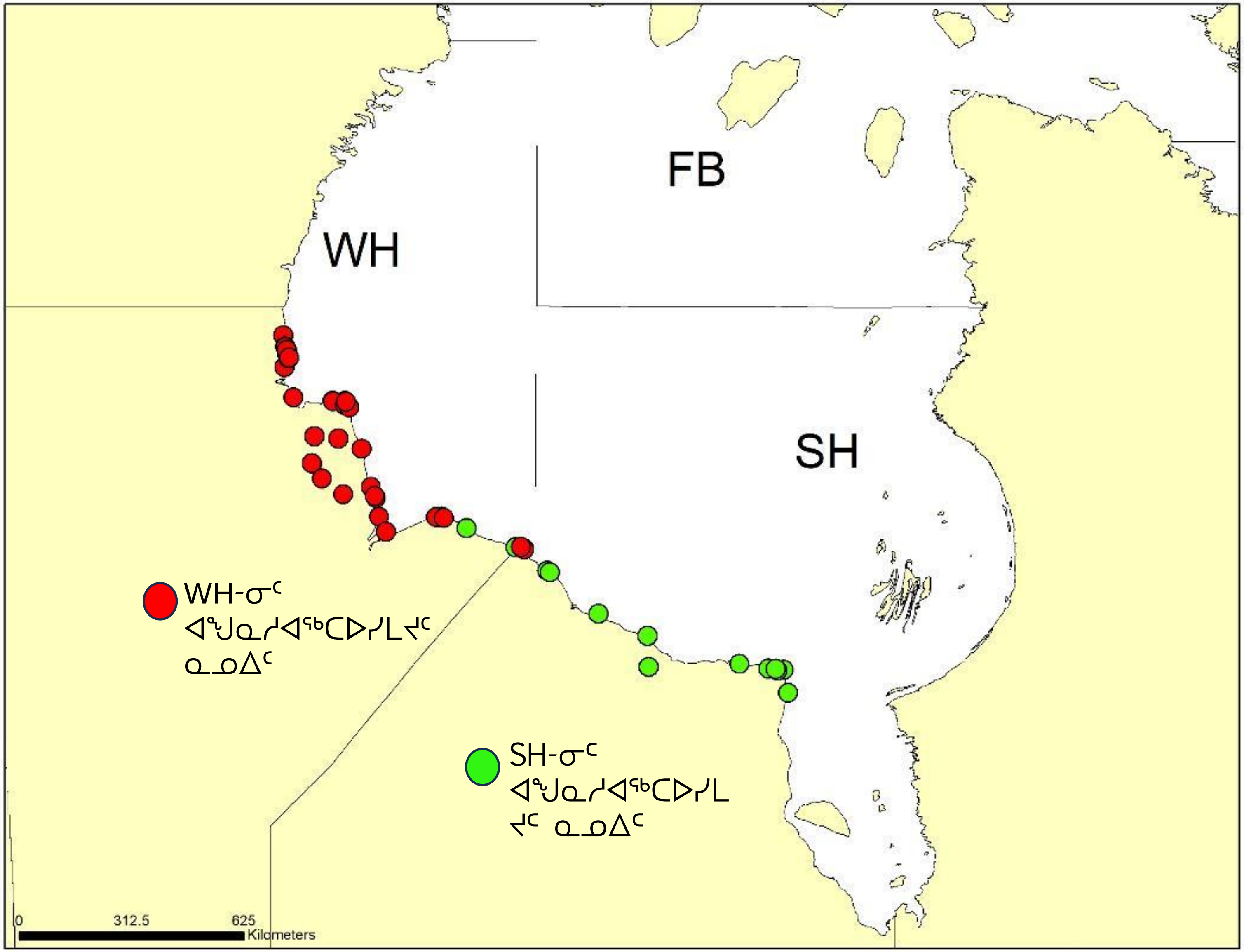


ᖃᓄᐃᓕᖅᓂᖕᒋᑦ - WH-ᓂᑦ SH-ᒧᑦ ᐊᒻᒪᓗ SH-ᓂᑦ WH-ᓂᑦ 2022/2023





WH-ᓂᑦ ᐊᒻᒪᓗ SH-ᓄᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᑎᒃᓴᓂᒃ ᐱᓯᒪᓕᓚᐅᖅᑐᑦ 2012-2023



🟥 WH-ᓂᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ

🟩 SH-ᓂᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ

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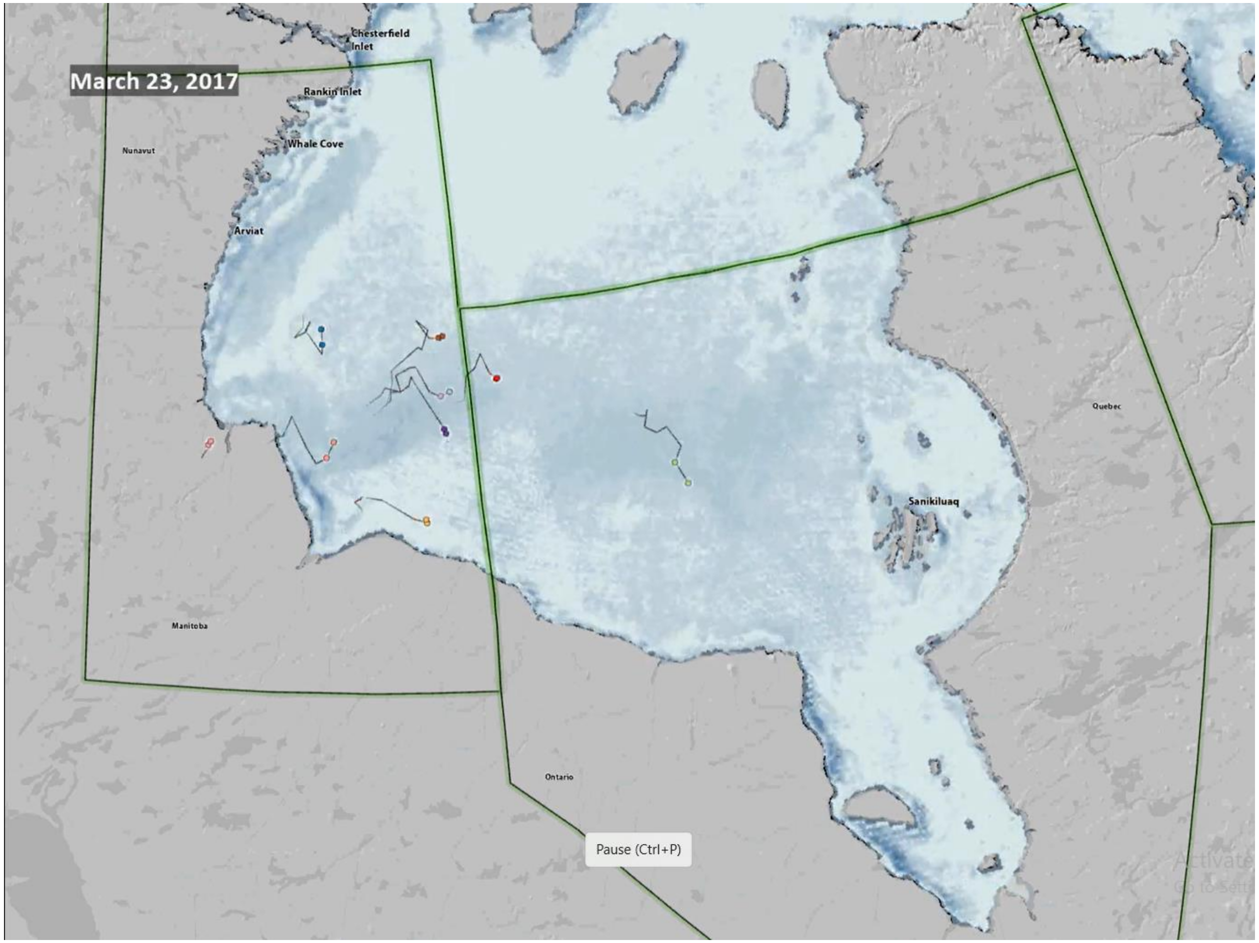
ᖁᖕᒐᓯᑭᖅᓯᓂᖅ



ᐱᓕᕆᔾᔪᓯᖅ ᐊᔾᔨᒌᖖᒋᓐᓂᖕᒐ

- ᖃᖕᒐᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ
    - "ᓯᓂᒃᓯᓂᖅ" ᑭᖕᒍᕙᖖᒋᕐᓗᓂ
    - ᖃᐅᔨᔪᓐᓇᖖᒋᓐᓂᖅ ᓇᓄᐃᑦ ᓇᒧᖖᒐᐅᕙᖕᓂᖕᒋᓐᓂᒃ

- ᐲᖅᓯᓂᖅ ᖃᐅᔨᓴᖅᑕᐅᔪᒃᓴᒥᒃ ᓯᓂᒃᑎᑕᐅᓗᑎᒃ
    - ᓇᓗᓇᐃᖅᓯᔪᓐᓇᖅᑐᑦ ᓇᓄᕐᓂᒃ
    - ᖃᐅᔨᒪᓕᕐᓗᒋᑦ ᐃᓚᖕᒋᑦ ᓇᓄᐃᑦ ᓇᒧᖖᒐᐅᕙᓪᓕᐊᕙᖕᓂᖕᒋᑦ

- ᕐᑯᖕᒐᓯᕈᖅᑕᐅᔪᑦ
    - ᓇᓗᓇᐃᖅᓯᔪᓐᓇᖅᑐᑦ ᓇᓄᕐᓂᒃ
    - ᖃᐅᔨᒪᓕᕐᓗᒋᑦ ᐊᐅᓚᓂᓪᓚᑦᑖᕆᔭᐅᕙᒃᑐᑦ 1-2–ᓄᑦ ᐊᕐᕌᒍᓄᑦ

ᖃᐅᔨᓴᖅᑕᐅᖏᓐᓇᖅᑐᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ

ᐊᕐᕕᐊᑦ

- ᔫ ᓴᕕᑲᑖᖅ ᓄᑲᖅᓚᖅ – ᐆᒪᔪᕐᓂᐊᖅᑎ

- ᕿᒫᑎᑦᑎᔾᔪᑎᑦ/ᑭᒃᑯᑐᐃᓐᓇᓂᒃ ᐊᑦᑕᕐᓇᖅᑕᐃᓕᒪᑎᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖅ

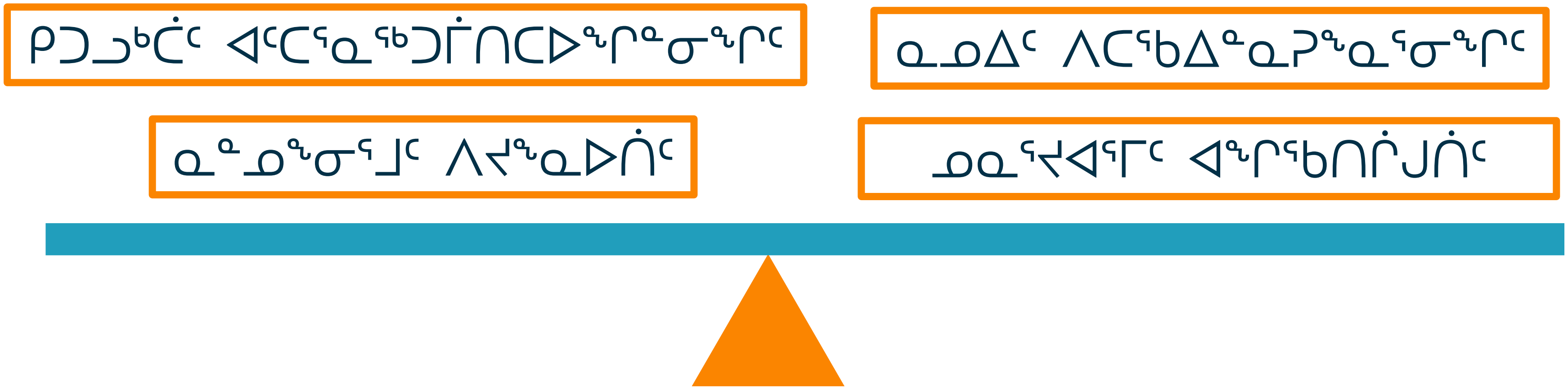
- ᐋᖅᓯᓂᖅ ᖃᐅᔨᓴᖅᑕᐅᔪᒃᓴᒥᒃ ᓯᓂᒃᑎᑕᐅᓗᑎᒃ
    - ᖃᔅᓯᑦ ᓇᓄᐃᑦ ᑖᒃᑯᐊᑦᑕᐃᓐᓇᐅᓪᓗᑎᒃ ᓇᓄᐃᑦ ᐅᑎᖅᐸᑉᐸᑦ?
    - ᖃᑦᑎᑦ ᓇᓄᐃᑦ ᐊᐳᖅᑕᕈᑎᖃᖅᐸᑦ ᑰᔾᔪᐊᕌᓗᒻᒥᑦ?

- ᑭᓱᑦ ᕿᒫᑎᑦᑎᔾᔪᑎᑦ ᐊᑑᑎᖃᓛᖕᒍᕙᑦ?
    - ᕐᑯᐊᖅ ᓇᑦᑎᕕᖕᓄᑦ ᐃᓂᒋᔭᐅᔪᑦ
    - ᕿᔭᖃᑦᑕᓕᕆᓂᖅ ᐊᒻᒪ ᓅᑦᑎᓂᖅ
    - ᕐᑯᐱᕐᕆᓯᒪᔪᑦ, ᓂᖀᑦ, ᐊᓯᖏᓪᓗ.

ᐱᓕᕆᐊᖑᓂᐊᓕᕐᒥᔪᑦ – ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ

ᐱᓕᕆᐊᖕᒐᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ

- ᐃᓪᓗᒌᖕᒐᓕᕆᓂᖅ



ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ (TAH)

- ᒫᓐᓇᐅᔪᖅ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐱᖕᒐᖕᓇᖅᐸᓱᖕᒐᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ 38 ᓇᓄᐃᑦ ᐅᑭᐅᑕᒫᑦ.
- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᒫᓐᓇ ᐊᑐᖁᔨᖕᒪᑕ **ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ 38-ᓂᑦ.**
  - ᑕᒪᓐᓇ ᑕᐃᒪᐃᔾᔪᑎᖃᖅᐳᖅ ECCC-ᒧᑦ ᑎᒥᖕᒋᓐᓂᑦ ᐲᖅᓯᓂᕐᒧᑦ ᖃᓄᐃᓕᖕᒐᓕᖅᑎᑕᐅᔪᓄᑦ **ᑕᑯᒃᓴᐅᑎᑦᑎᓪᓗᓂ ᓇᒧᙵᐅᕙᓪᓕᐊᓂᕋᔭᐅᔪᒥᒃ ᐅᑯᖕᓄᙵ, WH ᐊᒻᒪᓗ SH**

ᐱᓕᕆᐊᖑᓂᐊᓕᕐᒥᔪᑦ – ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ

- ᐊᖏᖃᑕᐅᕕᑦ ᐊᒥᓲᓂᖏᑦ ᓇᓄᐃᑦ **ᐊᑕᐅᓯᑦᑕᐃᓐᓇᓕᓂᖏᓐᓂᒃ** ᖃᑯᒍᙳᖅᐸᓪᓕᐊᑎᓪᓗᒍ?
- **ᓈᒻᒪᒃᐸᑦ** ᓇᓐᓄᒃᑕᐅᔪᒃᓴᑦ? **ᐊᒥᓲᙱᓗᐊᕐᐸᑦ**? **ᐊᒥᓲᓗᐊᖅᐸᑦ**?
- ᑭᓱᒥᓪᓗ ᐅᔾᔨᕈᓱᓚᐅᖅᐱᑦ ᓇᓅᑉ **ᑎᒥᐊᑕ ᖃᓄᐃᓐᓂᖓᓂ** ᖃᑯᒍᙳᖅᐸᓪᓕᐊᑎᓪᓗᒍ?
- ᐊᔾᔨᐅᙱᑦᑐᒥᑦ ᐅᔾᔨᕈᓱᓚᐅᖅᐱᑦ ᑐᓴᖅᑎᑦᑐᒪᓪᓗᑕᓗ?
- ᓇᐅᒃᑯᓪᓗ **ᐊᖏᖃᑎᖃᕐᐱᑦ/ᐊᖏᖃᑎᖃᙱᓚᑎᑦ** ᖃᐅᔨᔭᑦᑎᓐᓂᑦ?

ᖁᔭᓐᓇᒦᒃ!

ᖃᐅᔨᒋᐊᕐᕕᒃᓴᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᑦ:

ᐋᓖᓴ ᐴᕼᐋᑦ

✉ abohart@gov.nu.ca

ᐃᕖᓐ ᕆᑦᓱᑦᓴᓐ

✉ evan.richardson@ec.gc.ca

ᐊᐱᖅᑯᑎᒃᓴᖃᖅᐲᑦ?

ᐅᕐᑲᕐᑲᑎᕐᑲᕐᓂᑦ ᑲᖕᒋᖅᖠᓂᕐᒥᑦ, ᐊᕐᕕᐊᓂᕐ, ᐊᒻᒪᓗ ᐃᓴᑎᒃ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓂᑦ 2021-ᒥᑦ ᕐᑲᖕᒐᑕᓲᒃᑯᑦ ᕐᑲᐅᔨᓴᐃᖕᒪᑕ ᐱᖕᒐᖕᓇᖅᐸᓯᖕᒐᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒥᑦ ᓄᓇᐃᑦ ᐊᒥᓲᑎᒋᓂᖕᒋᓐᓄᑦ ᐅᓂᑉᑳᒥᑦ

ᓇᕐᑎᐊᑦ 26, 2024

ᓇᕐᑎᐊᑦ 27, 2024

ᓄᕐᕋᐃᑦ 9, 2024

ᓄᕐᕋᐃᑦ 10, 2024

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ
ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ
ᐃᕐᑲᓗᐃᑦ, ᓄᓇᕗᑦ

ᐊᖓᔪᖅᑳᖑᔪᓂᑦ ᓇᐃᓈᖅᓯᒪᔪᖅ

ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ, ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᑭᒡᒐᖅᑐᐃᔪᑦ ᑕᒡᕙᐅᖃᑎᒋᑉᓗᒋᑦ ᐅᖃᖃᑎᖃᓚᐅᖅᑐᑦ ᑲᖏᖅᖠᓂᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ ᓇᑦᑎᐊᑦ 27-ᒥᑦ ᐊᕐᕕᐊᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ ᓇᑦᑎᐊᑦ 27, ᐊᒻᒪᓗ ᐃᓴᑎᒃ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ ᓄᕐᕋᐃᑦ 9-ᒥᑦ ᐊᒻᒪᓗ 10-ᒥᑦ, 2024-ᒥᑦ.

ᐱᔾᔪᑎᒋᔭᖓ ᐅᖃᖃᑎᖃᕐᓃᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᒻᒪᓗ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐅᖃᐅᓯᖃᕈᖕᓇᕐᓂᐊᕐᓗᑎᒃ ᖃᓄᐃᖓᓂᐅᔪᓂᒃ 2021 ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᐃᖕᒪᑕ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓄᑦ ᐅᓂᑉᑳᒥᑦ, ᑲᒪᒋᔭᐅᑉᓗᓂ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓄᑦ. ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᑲᑎᑕᐅᔪᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ 2021-ᒥᑦ. ᓯᕗᓂᐊᓂᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓄᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᑲᑎᑕᐅᓚᐅᖅᑐᑦ 2011-ᒥᑦ ᐊᒻᒪᓗ 2016-ᒥᑦ. 2021-ᒥᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᖃᓄᐃᖓᓂᐅᔪᑦ ᐅᖃᖅᑐᑦ ᐊᒥᓲᔪᖕᓃᖅᓯᒪᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ 2016-ᒥᑦ 2021-ᒧᑦ, ᑭᓯᐊᓂ ᖃᓄᐃᖓᓂᐅᔪᑦ ᑲᑎᑕᐅᓚᐅᖅᑐᑦ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᐃᓂᐅᓚᐅᖅᑐᓂᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓄᑦ ᖃᓄᐃᖓᓂᐅᔪᓂᑦ, ᐊᔾᔨᒌᙱᓐᓂᖃᓚᐅᖅᑐᑦ ᐊᑕᐅᓯᕐᒥᑦ ᓇᓄᕐᒥᑦ ᐊᑯᓐᓂᖓᓂᑦ 2016 ᐊᒻᒪᓗ 2021. ᑭᒡᓕᖃᕐᓂᖓᓄᑦ ᐱᔾᔪᑎᒥᑉᓗᒍ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᓂᒃ, ᖃᐅᔨᔭᐅᔪᖕᓇᓚᐅᖏᑦᑐᖅ ᖃᓄᑎᒋᐅᔪᒥᒃ ᐊᒥᓲᔪᖕᓃᖅᒪᖔᑕ ᐊᐅᓚᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ. ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ (ECCC) ᖃᐅᔨᓴᐃᖃᑦᑕᖅᓯᒪᔪᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᑕᐃᒪᙵᓂᑦ 1980-ᖏᓐᓂᑦ, ᐃᓚᐅᑉᓗᑎᒃ ᓂᕿᖏᓐᓂᒃ ᐲᖅᓯᕝᕕᐅᑉᓗᑎᒃ 2017-ᒥᑦ ᐅᑉᓗᒥᒧᑦ ᑎᑭᖦᖢᒍ. ᖃᓄᐃᖓᓂᐅᔪᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ ᐅᖃᖅᑐᑦ ᐊᒥᓲᔪᖕᓃᖅᓯᒪᓂᖏᑦ ᐃᓚᖓᒍᑦ ᐱᔾᔪᑕᐅᔪᖅ ᐊᐅᓚᓐᓂᖏᓐᓄᑦ ᖃᓂᑖᓂᑦ ᑭᒡᓕᒋᔭᐅᔫᑉ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ 2021-ᒥᑦ, ᐊᒥᓲᔪᖕᓃᖅᓂᕐᒥᐅᖏᑦᑐᖅ ᐊᓐᓇᒃᑐᖅᓯᒪᔪᖕᓇᖅᓂᖏᑦ ᐱᔾᔪᑕᐅᑉᓗᓂ. ᑐᕌᖓᓇᔭᐅᔪᖅ ᐅᖃᖃᑎᖃᕐᓂᕐᒧᑦ ᐊᒻᒪᓗ ᑕᑯᔭᒃᓴᐅᑎᒋᔭᐅᔪᓄᑦ ᑐᓂᓯᓂᐊᕐᓗᑎᒃ ᐊᐅᓚᑦᑎᖃᑎᒋᔭᐅᔪᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᖃᓄᐃᖓᓂᐅᔪᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐅᓂᑉᑳᖓᓐᓂᑦ ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ ᖃᐅᔨᓴᐃᓂᖓᓐᓂᑦ, ᑲᑎᑦᑎᓗᑎᒃ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐅᖃᐅᓯᐅᔪᓂᑦ ᖃᓄᐃᖓᓂᐅᔪᓄᑦ, ᑲᑎᑦᑎᒃᑲᓐᓂᕐᓗᑎᒃ *ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᑦ* (IQ), ᐊᒻᒪᓗ ᐅᖃᐅᓯᕆᓗᒋᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑲᑎᑐᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ (TAH) ᐃᒪᓈᖅᑐᑕᐅᔪᑦ.

ᓇᓄᓕᕆᔩᑦ ᑲᑎᖃᑎᖃᓚᐅᖅᑐᑦ ᐊᑐᓂ ᓄᓇᓕᖕᒥᑦ ᑭᒡᒐᖅᑐᐃᔪᓂᒃ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ ᓇᓄᖅᓯᐅᖃᑦᑕᖅᑐᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ.

ᐃᑲᔪᖅᑐᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᖏᓐᓄᑦ ᖃᐅᔨᑎᑦᑎᓚᐅᖏᑦᑐᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᐅᖃᖃᑎᒋᔭᐅᖕᒪᑕ. ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐅᖃᓚᐅᖅᑐᑦ ᑕᑯᔪᒪᓂᖅᓴᖅᓗᑎᒃ ᐊᒥᓱᒻᒍᓄᐊᖅᑕᐅᖁᑉᓗᒍ ᑲᑎᑐᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ. ᐅᖃᑦᑐᓇᒃᓴᐅᑎᐊᓂᖕᒪᑕ, ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐊᐱᖅᑯᑎᒃᓴᖃᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ ᐃᓱᒪᒋᔭᖏᓐᓂᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᐅᕙᓘᓐᓃᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ (NWMB), ᓇᓕᐊᑐᐃᓐᓇᖅ ᐱᔪᒪᔭᖓᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ.

ᐃᓱᒪᒋᔭᐅᔪᑦ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᑲᑎᑕᐅᓚᐅᖅᑐᑦ ᐅᖃᖃᑎᒌᒃᑎᑦᑐᒥ ᓇᐃᓈᖅᑎᖅᑕᐅᓂᐊᖅᑐᑦ ᑐᓂᔭᐅᒃᐸᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓄᑦ ᐃᒪᓈᖅᑑᑕᐅᔪᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᖏᓐᓄᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᓂᐊᖅᑐᑦ.

ᐅᓂᑉᑳᖅ ᓇᐃᓈᖅᑎᕆᓇᓱᒃᑐᖅ ᐅᖃᐅᓯᐅᓚᐅᖅᑐᓂᒃ ᐃᓚᐅᖃᑕᐅᔪᓄᑦ ᐅᖃᖃᑎᓚᐅᑎᒡᓗᒋᑦ.

ᑐᑭᓯᑎᑦᑎᓂᖅ

ᐅᓂᑉᑳᒥᑦᑐᑦ ᐊᑭᑎᓕᕆᔨᒃᑯᑦ ᐱᑦᑕᐅᓛᒃᑯᑦ ᐱᓕᕆᐊᕆᔭᖏᑦ ᓇᓚᐅᑦᑎᔪᒃᑯᑦ ᐱᓂᐊᕐᓗᒋᑦ ᑕᒪᕐᒥᓗᒃᑖᖅ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᑐᓂᔭᐅᓚᐅᖅᑐᑦ ᑲᑎᒪᑎᑦᑎᑎᓪᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓂᑦ ᑲᖏᖅᖠᓂᕐᒥᑦ ᓇᑦᑎᐊᑦ 26-ᒥᑦ ᐊᕐᕕᐊᓂᑦ ᓇᑦᑎᐊᑦ 27-ᒥᑦ, ᐊᒻᒪᓗ ᑎᑭᕋᕐᔪᐊᕐᒥᑦ ᓄᕐᕋᐃᑦ 9-ᒥᑦ ᐊᒻᒪᓗ 10-ᒥᑦ, 2024-ᒥᑦ.

ᑕᐅᑐᙵᔭᐅᔪᑦ ᖃᐅᔨᑎᑦᑎᔾᔭᑕᐅᓚᐅᖅᑐᑦ ᑕᒡᕙᓂ ᑕᑯᒃᓴᐅᑎᑦᑎᓗᑎᐅᖏᑦᑐᖅ ᐊᑭᑎᓕᕆᔨᒃᑯᓐᓂᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓂᑦ.

ᐃᓗᓕᖏᑦ

ᐃᓗᓕᖏᑦ

1.0 ᐅᓂᑉᑳᑉ ᐱᔾᔪᑎᖓ ᐊᒻᒪᓗ ᐋᖅᑭᐅᒪᓂᖓ

ᐅᓂᑉᑳᖅ ᐋᖅᑭᒃᓯᐃᓂᐅᔪᖅ ᐊᒻᒪᓗ ᓇᐃᓈᖅᑎᓇᔪᖅ ᐅᖃᐅᓯᐅᔪᓂᑦ, ᐊᐱᖅᑯᑕᐅᔪᓂᑦ, ᐃᓱᒫᓘᑕᐅᔪᓂᑦ ᐊᒻᒪᓗ ᐱᖅᑯᔨᓂᖕᒋᓐᓂᑦ ᐱᔭᐅᔪᑦ ᑲᖕᒋᖅᖠᓂᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ, ᐊᕐᕕᐊᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ, ᐊᒻᒪᓗ ᐃᓴᑎᒃ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᖃᓄᐃᖓᓂᐅᔪᓂᒃ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᐃᓂᕐᓂᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᕐᓄᑦ ᐅᓂᑉᑳᒥᒃᒃ.

ᑭᒡᒐᖅᑐᐃᔪᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᑦ ᐅᖃᖃᑎᖃᖅᑐᓃᖃᑕᐅᓚᐅᖅᑐᑦ ᐊᓯᖏᓐᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᖃᑎᒋᔭᐅᔪᓂᑦ ᐊᒻᒪᓗ ᐱᔪᒪᓂᖃᖅᑐᓂᑦ ᐃᓚᒋᔭᐅᔪᓂᑦ.

2.0 ᐱᔾᔪᑎᖓ ᐅᖃᖃᑎᖃᕐᓂᐅᑉ

ᐱᔾᔪᑎᒋᔭᖓ ᐅᖃᖃᑎᖃᕐᓂᐅᑉ ᐅᖃᐅᓯᕆᓂᐊᕐᓗᒋᑦ ᓄᑕᐅᙳᐊᑦ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒧᑦ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᐱᔾᔪᑎᖃᖅᑐᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ ᐅᖃᐅᓯᐅᓂᖓᑐᑦ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓄᑦ ᓈᓴᖅᑕᐅᔪᓄᑦ ᐅᓂᑉᑳᒥᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᒃᑯᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ. ᐃᓚᒋᔭᐅᑉᓗᓂ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑐᓂᓯᓚᐅᕐᒥᔪᑦ ᑲᑎᓗᒃᑖᖅᓗᑎᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ ᐊᑐᖅᑕᐅᖅᑯᔭᖏᓐᓂᑦ ᐅᖃᖃᑎᖃᖅᑎᓪᓗᒋᑦ, ᑭᓯᐊᓂ ᐅᖃᐅᓯᖅᑲᓚᐅᕐᒥᔪᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᐅᔪᑦ ᐋᖅᑭᒃᓯᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᑐᖅᓗᒋᑦ ᓄᓇᓖᑦ ᐱᔭᓇᐊᖅᑲᖅᑕᖏᑦ ᐊᒻᒪᓗ ᐱᔭᒃᓴᐅᑎᖏᑦ ᓇᓄᕐᓄᑦ.

3.0 ᐋᖅᑭᐅᒪᓂᖓ ᑲᑎᒪᑎᑦᑎᓂᐅᑉ

ᑲᑎᒪᖃᑦᑕᓚᐅᖅᑐᑦ ᐅᓐᓄᒃᓴᒃᑯᑦ ᐊᒻᒪᓗ ᐊᑕᐅᓰᖅᓗᑎᒃ ᐅᐸᑦᑳᒃᑯᑦ, ᐊᒻᒪᓗ ᒪᕿᖕᓂᑦ ᐱᖓᓱᓄᑦ ᐃᑲᕐᕋᓄᑦ ᑲᑎᒪᖃᑦᑕᕐᓗᑎᒃ. ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᓇᓄᓕᕆᔨᖓ, ᐊᓕᓴ ᐳᕼᐊᑦ, ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ ᓇᓄᓕᕆᔨ, ᐊᕙᓐ ᓇᑦᓯᑦᓴᓐ ᑲᒪᔭᖅᑲᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᑲᑎᒪᑎᑦᑎᓪᓗᓂ. ᐊᑐᓂ ᑕᑯᔭᒃᓴᐅᑏᑦ ᐱᒋᐊᓚᐅᖅᑐᑦ ᐅᒃᑯᐃᖅᓯᓂᕐᒧᑦ ᐅᖃᓪᓚᒃᑐᓂᑦ ᐊᓕᓴ ᐳᕼᐊᑦ ᐊᒻᒪᓗ ᐊᕙᓐ ᓇᑦᓯᑦᓴᓐ ᑭᓇᐅᓂᖏᓐᓄᑦ ᖃᐅᔨᑎᑦᑎᑉᓗᑎᒃ ᐊᒻᒪᓗ ᓇᓕᐊᖕᓂᑦ ᐊᐅᓚᑦᑎᔨᓂᙵᓖᖅᒪᖕᓖᑕ ᑭᒡᒐᖅᑐᖅᑕᖏᓐᓂᑦ ᐊᒻᒪᓗ ᐱᔾᔪᑎᒋᔭᖓᓂᑦ ᐅᖃᖃᑎᖃᕐᓂᐅᑉ. ᑭᖑᓂᐊᒍᑦ ᑕᖅᓇᔭᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᐱᔾᔪᑎᖃᖅᑐᒥᒃ ᖃᐅᔨᓇᓱᖕᓂᒃᑯᑦ ᓈᓴᐃᓂᕐᒥᒃ ᖃᐅᔨᓴᐃᓂᕐᒥᒃ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᔨᐅᖅᑲᑕᐅᓂᖏᓐᓄᑦ ᐱᓕᕆᐊᖑᔪᒥᑦ, ᐊᐅᓚᑦᑎᓂᐅᕙᓚᐅᖅᑐᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ, ᓇᑐᓇᐃᔭᖅᓯᒪᔪᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᑦ ᖃᓄᐃᖓᓂᐅᔪᓄᑦ 2021-ᒥᑦ ᓇᓄᕐᓂᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ, ᐊᒻᒪᓗ ᖃᓄᐃᖓᓂᐅᔪᓂᒃ ᖃᐅᔨᓇᓱᖕᓂᒃᑯᑦ ᐱᓕᕆᐊᖑᔪᓂᒃ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ. ᐱᓕᕆᐊᖑᔪᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ ᐃᑲᔪᓚᐅᖅᑐᑦ ᐅᖃᐅᓯᕆᑉᓗᒍ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᖃᓄᐃᖓᓂᐅᔪᓄᑦ. ᐃᓚᐅᖃᑕᐅᔪᑦ ᐊᐱᖅᑯᑎᒃᓴᖃᕈᑎᒃ ᐊᐱᖅᓱᖅᑯᔭᐅᓚᐅᖅᑐᑦ, ᖃᐅᔨᑎᑦᑎᓗᑎᒃ ᐃᓱᒫᓘᑎᒋᔭᖏᓐᓂᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᓱᖁᔨᓂᖏᓐᓂᒃ ᑲᑎᒪᑎᓪᓗᒋᑦ. ᑲᑎᒫᓂᒃᑎᓪᓗᒋᑦ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᑐᖅᑕᐅᖁᔭᖅᓯᒪᕗᑦ ᑲᑎᒪᖃᑦᑕᕐᓗᑎᒃ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑐᓂᔭᐅᓪᓗᑎᒃ.

4.0 ᓇᐃᓈᖅᓯᒪᔪᑦ ᐅᖃᓪᓗᓇᒃᓴᐅᑎᓂᕐᓂᑦ

ᓇᐃᓈᖅᓯᒪᔪᑦ ᑲᑎᑕᐅᓚᐅᖅᑐᑦ ᐱᔾᔪᑕᐅᔪᓄᑦ ᐊᒻᒪᓗ ᐃᓱᒪᒋᔭᐅᔪᓄᑦ ᐱᔭᐅᓚᐅᖅᑐᓂᑦ ᐱᖓᓱᓂᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᖑᔪᓂᑦ. ᓇᐃᓈᖅᓯᒪᔪᓄᑦ ᐅᖃᐅᓯᐅᓗᐊᓚᐅᖅᑐᓂᑦ ᐊᒻᒪᓗ ᐃᓚᐅᖃᑕᐅᓚᐅᖅᑐᓂᑦ ᐊᑐᓂ ᐅᖃᖃᑎᒋᔭᐅᔪᓂᑦ, ᑕᑯᓗᒍ ᑎᑎᖅᖃᒧᑦ ᐃᓚᓕᐅᑎᓯᒪᔪᖅ A.

<u>ᐅᖃᐅᓯᐅᔪᑦ ᐊᒻᒪᓗ ᐊᐱᖅᖁᑏᑦ:</u>

- ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐅᖃᖅᑐᑦ ᑕᑯᔪᒪᔪᑦ ᐊᒥᓱᙳᕈᐊᖅᑕᐅᓗᑎᒃ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ. ᐅᖃᓪᓗᓇᒃᓴᓚᐅᖅᑐᑦ ᖃᐅᔨᓂᐊᕐᓗᑎᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᓂᒃ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᓂᖏᓐᓂᑦ.
- ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᖃᐅᔨᑎᑦᑎᓚᐅᖅᑐᑦ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᑦᑐᖕᓇᕐᓂᖏᓐᓄᑦ ᐃᓱᒪᓘᑎᓂᒃ:
  - ᑐᐱᖅᓯᒪᓂᖅ ᐊᔪᕐᓇᖅᓯᔪᖅ, ᐱᑭᐅᑕᓇᐊᕐᓂᖅ, ᖁᑭᐅᑎᓂᒃ ᓇᒃᓴᐃᓐᓇᖅᑲᑦᑕᓇᐊᖅᑲᑦᑕᓕᖅᑐᑦ, ᓇᓄᐃᑦ ᐃᖃᓗᖕᓂᑦ ᒪᑦᑎᑕᐅᑎᓂᒃ ᐱᖃᑦᑕᖅᑐᑦ, ᓇᑦᑎᓕᓇᓂᖅ ᕿᓇᓗᒐᕐᓂᓗ ᐊᔾᔨᒋᔪᙳᖅᑕᖅᑐᑦ ᓇᓄᐃᑦ ᑎᑭᑎᑦᑕᐃᓕᓂᐊᕐᓗᒋᑦ, ᐊᒻᒪᓗ ᓇᓄᐃᑦ ᐃᒡᓗᕐᔪᐊᕋᓛᓂᑦ/ᐱᖁᑎᒋᔭᐅᔪᓂᒃ ᓯᕿᐃᖃᑦᑕᖅᑐᑦ.
  - ᖃᑉᓯᐅᓂᖏᑦ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓯᒪᑎᓪᓗᒋᑦ ᐊᒻᒪᓗ ᑐᖁᑕᐅᔪᖃᖅᓱᓂ ᐃᓱᒪᒋᔭᐅᔭᕆᐊᖃᖅᑐᑦ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᔪᓂᑦ.
- ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᖃᐅᔨᑎᑦᑎᓚᐅᖅᑐᑦ ᓈᒻᒪᒃᓴᙱᓐᓂᖏᓐᓂᒃ ᐊᐅᓚᑦᑎᔨᐅᖅᑕᐅᓂᖓᓄᑦ ᐋᖅᑭᐅᒪᔪᒥᒃ ᐊᒻᒪᓗ ᖃᐅᔨᑎᑦᑎᑉᓗᑎᒃ ᐃᓱᒪᒋᔭᖏᑦ ᑐᓵᔭᐅᒃᑖᙱᒻᒪᑕ ᐊᒻᒪᓗ ᐅᖃᖃᑎᒋᓯᒪᙱᒻᒪᒋᑦ ᑲᑎᒪᔩᑦ ᓯᕙᓕᑭᐊᙳᔾᔪᓯᓇᔭᐅᑉᓗᑎᒃ. ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐃᓚᐅᑎᑕᐅᓂᖅᓴᐅᔭᕆᐊᖃᖅᑐᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐋᖅᑭᐅᒪᔪᓂᑦ. ᐱᔾᔪᑕᐅᔪᒥᒃ ᖃᐅᔨᓴᐃᓂᖅᑕᖃᕈᓐᓇᖅᑐᑦ ᑲᑎᑦᑎᓂᐊᕐᓗᑎᒃ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᒃ ᐃᓱᒪᓕᐅᕆᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓂᐊᕐᓗᑎᒃ. ᓈᒻᒪᒃᓴᙱᑦᑐᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐃᓱᒪᓕᐅᕆᓂᐊᕐᓂᖅ ᐊᑐᖅᑕᐅᓯᒪᓂᖓ ᐃᓄᐃᑦ ᐃᓅᓯᖓᓂᑦ.
- ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐅᖃᖅᑐᑦ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᒧᑦ ᐋᖅᑭᐅᒪᔪᖅ ᐋᖅᑭᐅᒪᙳᖅᑐᖅ ᐃᓄᐃᑦ ᐱᖅᑯᓯᖏᓐᓄᑦ:
  - ᐱᒻᒪᕆᙳᖅᑕᖅᑎᑦᑎᖅᑕᖅᑐᖅ ᐊᓯᙳᓂᑦ ᓄᓇᓕᙳᓂᑦ, ᑕᒪᓐᓇ ᐃᓄᐃᑦ ᐱᖅᑯᓯᖓᙳᖅᑕᙳᓚᑦ.
  - ᓇᓐᓄᓗᐊᖅᑕᖅᑕᖅᑐᙳᖅᑐᑦ ᐃᓄᐃᑦ ᐊᖑᑎᑉᓗᒍ, ᐱᔾᔪᑎᑉᓗᒍ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᒥᓲᙳᖅᓱᒍᑦ ᐊᒥᓱᙳᕈᐊᖅᑐᒋᑦ. ᓇᓄᖅᓯᐅᕐᓂᖅ ᐱᓕᕆᐅᖅᑕᐅᔪᖅ ᐃᓚᒋᔭᐅᑉᓗᓂ ᐃᓄᐃᑦ ᐱᖅᑯᓯᖑᓂᑦ ᐊᒻᒪᓗ ᐊᖑᑎᙳᖅᐸᓪᓕᐊᓂᖓᓄᑦ.
  - ᐃᒃᐱᒍᓱᙱᓐᓂᖅᑐᖅ "ᐃᒃᐱᒍᓱᒃᑲᐅᓂᖅᑖᕆᓗᐊᖅᓂᖅᑎᑦ ᐃᓐᓇᕐᔭᐅᔪᑦ ᐊᒻᒪᓗ ᐱᖅᑯᓯᖅᑎᑦ".
- ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᖃᐅᔨᑎᑦᑎᔪᑦ ᐃᓱᒫᓗᑎᒋᔭᖅᑎᓂᑦ ᑭᓪᓕᖅᑐᓂᒃ/ᓇᓚᐅᑦᑎᓯᒪᓂᖅᒥᒃ ᑎᖅᒥᓲᒃᑯᑦ ᓈᓴᐃᖅᑕᖅᑎᑦᑐᑎᑦ (ᐃᒫᓃᑦᑐᓂᑦ ᓄᓇᖅᓂᑦ ᐱᖅᑕᖅᑎᑦᑐᑦ, ᑕᑯᓴᐅᑎᑦᐊᖅᓇᖅᑎᑦᑐᑦ ᐱᔾᔪᑕᐅᔪᑦ). ᐅᒃᐱᕆᔭᖅᑲᓗᐊᖅᑎᓂᖅ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖅᒥᑦ.
- ᐃᓱᒪᖅᓱᕐᓂᖏᔭᐅᔪᓂᒃ ᖃᐅᔨᑎᑦᑎᓚᐅᖅᑐᑦ ᓄᓇᖅᓂᑦ ᑕᑯᔭᖅᑐᐃᖅᑕᖅᑐᖅᑐᓂᒃ ᑰᓪᔪᐊᖅᒥᑦ ᐊᒻᒪᓗ ᑭᑐᑐᐃᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᖏᓐᓂᒃ ᓴᖅᑭᑎᓂᖕᒐ ᓄᓇᖕᒥᐅᓄᑦ, ᐃᓚᐅᑉᓗᑎᒃ ᐃᓂᒋᔭᐅᔪᓂᒃ ᓇᓄᖅᓄᑦ, ᑐᓴᑎᐊᕈᓐᓃᖅᑎᑦᑎᖅᑕᖅᑐᑦ ᐊᑐᖅᑕᐅᓂᖅᑐᑦ ᓂᐱᖅᑯᖅᑐᑦ ᖄᒥᑎᑎᓇᓱᒃᑐᑎᒃ, ᓂᖅᑭᖅᑎᑦ ᓯᖅᑕᐅᔪᑦ ᕈᐊᐳᐃᑦ ᖃᔭᐃᑦ ᓂᖅᑭᖅᑎᓄᑦᓇᖅᓯᒪᔪᓄᑦ, ᐊᒻᒪᓗ ᐊᖅᑲᐃᑦ ᖃᓂᖃᓄᑦᓇᕐᑕᐅᖅᑲᑦᑕᖅᓂᖅᑎᑦ.
- ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐊᐱᖅᓱᖅᑐᑦ ᐃᓚᒌᖕᒐᓂᖅᑎᓂᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓐᓂᖅᓯᖅᑎᓂᑦ ᐱᙳᓇᖅᐸᓯᖕᒐᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ, ᓂᕆᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ, ᐊᒻᒪᓗ ᑕᓯᐅᔭᕐᒥᑦ. ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐊᐱᖅᓱᖅᑐᑦ ᖁᖓᓯᖃᓯᖅᑐᐃᓂᖅᒥᒃ ᐊᒻᒪᓗ ᓂᖅᑭᖅᑎᑦ ᐲᖅᓯᖅᑕᖅᑎᓂᑦ ᐊᑐᖅᑕᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐊᑦᑕᕐᓇᖅᑐᖅᑕᖅᑐᕐᓂᖅᑎᑦᑐᒦᑎᑕᐅᓂᖅᑎᑦ ᓇᓄᐃᑦ.

ᑎᑎᖅᖃᒧᑦ ᐃᓚᓕᐅᑎᓯᒪᔪᖅ A

2021-ᒥᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐅᓂᑉᑳᑉ ᒥᒃᓵᓄᑦ ᐅᖃᖃᑎᖃᕐᓂᕐᒧᑦ ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᑲᖏᖅᖠᓂᖅ

ᐃᑲᓂ: ᑲᖏᖅᖠᓂᖅ, ᓄᓇᕗᑦ, ᑲᖏᖅᖠᓂᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᑎᑎᕋᕝᕕᖓᑦ
ᐅᑉᓗᖅ/ᓇᒧᖓᓕᑎᓪᓗᒍ: ᓇᑦᑎᐊᑦ 26, 2024
ᐱᒋᐊᕐᒪᑦ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 19:17
ᐃᓱᓕᑦᑐᖅ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 21:31

ᐃᓚᐅᖃᑕᐅᔪᑦ

ᑖᓇ ᐋᑕᒻᔅ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᑎᑎᕋᖅᑎ)
ᕼᐃᐅᓇ ᐃᑎᓐᓄᐊᖅ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐃᒃᓯᕙᐅᑕᖅ)
ᓃᕙᐃ ᑰᓕ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᑎᑎᕋᖅᑎ)
ᑯᓇᔅ ᑳᓂᓕ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᑲᑎᒪᔨ)
ᓴᕙᑲᑖᖅ ᖃᐅᒪᒃ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᑲᑎᒪᔨ)
ᓵᒻ ᐊᓚᕋᓚᒃ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᑲᑎᒪᔨ)
ᐊᓕᓴ ᐳᕼᐊᑦ, ᓇᓄᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ
ᐊᕙᓐ ᕆᑦᓯᑦᓴᓐ, ᓇᓄᓂᒃ ᐆᒪᔪᓕᕆᔨ, ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ
ᐊᔾᕈᐊ ᒍᕇᓐ, ᐃᓇᖐᓯᒪᔪᖅ ᐊᖓᔪᒃᖠᖅ ᐅᖃᐅᔾᔨᔨ – ᓄᓇᒥᒃ ᐊᑐᕐᓂᕐᒧᑦ ᐊᒻᒪᓗ ᒪᓕᒐᓂᒃ ᕿᒥᕐᕈᓂᕐᒧᑦ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ
ᕈᐃᒪᓐ ᒨᓯ, ᐱᕝᕙᐅᔪᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᑭᒧᐊᒃᑎᑦᑎᔨ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ
ᑭᓃᑕᓐ ᑕᖅᑕᖅ, ᑲᑎᒪᔫᑦᑐᖅ
ᔭᐊᓐ ᑰᑐ-ᐊᐅᑐᑦ, ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᐆᒪᔪᓕᕆᔨ ᑲᖏᖅᖠᓂᖅ
ᑖᓂᐅᓪ ᖃᓗᔾᔭᖅ, ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᐆᒪᔪᓕᕆᔨ ᑲᖏᖅᖠᓂᖅ

ᐱᔾᔪᑎᒋᔭᖓ ᑲᑎᒪᓂᐅᑉ:

ᑐᓂᓯᓗᑎᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓄᑕᐅᓛᓂᒃ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒥᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᖃᑉᓯᐅᑎᒋᓂᖏᓐᓄᑦ ᐅᖃᐅᓯᐅᓯᒪᓂᖓᑐᑦ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓄᑦ ᐅᓂᑉᑳᒥᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖏᓐᓂᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᑎᑐᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔭᓐᓇᖅᑐᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᖁᔭᓐᒋᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓯᓇᑐᐃᓐᓇᐃᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐊᐱᖅᖁᑎᒋᔭᐅᔪᑦ ᑕᒪᐃᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᓄᑦ. ᑲᑎᑦᑎᓗᓂ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᑭᖑᓪᓕᕐᒥᑦ ᐱᓕᕆᐊᖑᓂᐊᖅᑐᑦ ᐊᒻᒪᓗ ᖃᓄᐃᓕᐅᕈᑕᐅᒋᐊᕐᓂᐊᖅᑐᑦ:

ᓇᐃᑦᑐᒥᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (4-6 ᐱᓇᓱᐊᕈᓯᕐᓂᑦ ᑲᑎᒪᓚᐅᖅᖄᖅᑎᓪᓗᒋᑦ):

- ᖃᓄᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᓪᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐊᐱᖅᖁᑕᐅᔪᑦ
- ᖃᓄᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᓇᓗᓇᐃᕐᓗᒍ ᓇᓚᐅᑦᑎᔪᖅ ᑎᑎᕋᖅᓯᒪᓂᖓᑕ ᑎᖕᒦᒃᑯᑦ ᓅᓴᐃᖅᑕᐅᔪᑦ ᐊᑎᖏᑦ
- ᖃᓄᑕᐅᔭᒃᑯᑦ ᑎᑎᕋᕝᕕᒋᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᖃᓄᖅ ᐅᖓᓯᒃᑎᒋᔪᒥᑦ ᓄᓇᒥᑦ ᓅᓴᐃᓚᐅᕐᒪᖓᑕ
- ᖃᓄᑕᐅᔭᒃᑯᑦ ᑎᑎᕋᕝᕕᒋᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐳᓴᓐᑎᖏᑦ ᓇᓐᓄᒃᑕᐅᔪᑦ ᖁᓕᑦ ᐅᑭᐅᓂᐅᓕᖅᑐᒥᑦ

ᐊᑯᓂᐅᓂᓴᒧᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (2-3 ᑕᖅᕿᑦ ᐃᓗᐊᓂᑦ ᑲᑎᒪᓚᐅᖅᖄᖅᑎᓪᓗᒋᑦ):

- ᑐᓂᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐊᐅᓚᓐᓂᖏᓐᓂᒃ ᖁᖓᓯᔨᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ
- ᐱᔭᒪᓗᓂ ᑕᑯᔭᐅᖃᑦᑕᖅᑐᓂᒃ ᐅᓂᑉᑳᓂᑦ ᓇᓄᕐᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐊᐅᑖᖅᑎᑦᑎᔨᓂᑦ

ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ:

ᐊᐱᖅᖁᑏᑦ ᑲᑎᕐᒪᔪᓄᑦ ᑎᑭᓯᑎᑕᐅᖅᖄᖅᑎᓪᓗᒋᑦ ᑭᓲᖕᒪᖔᑕ:

ᑲᑎᒪᒋᐊᓚᐅᖅᑐᑦᖃᐅᔨᑎᑕᐅᑉᓗᑎᒃ ᑲᑎᒪᔫᖅᑕᐅᔪᓂᑦ. ᐊᓕᓴ ᐳᕼᐊᑦ ᐊᔾᔨᓂᒃ ᑕᐅᑐᒃᑎᑦᑎᔪᖅ ᐊᓯᐊᓄᐊᖅᐸᓪᓕᐊᑉᓗᑎᒃ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᐃᔨᒃᑯᑦ ᐃᓅᔪᖕᓃᓕᐅᕐᓂᖓᓄᑦ ᐆᒪᔪᓕᕆ ᒫᑲᔅ ᑎᒃ. ᑕᑯᑎᑦᑎᑕᖅᓱᓂ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᐃᓃᑦ ᐃᓚᐅᑉᓗᑎᒃ ᐸᕐᓇᐃᓂᖅᑳᖅᑎᓪᓗᒋᑦ ᑎᑭᖅᓱᒍ ᑲᑎᑐᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔭᓐᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ

ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᓱᒪᓕᐅᕆᓂᖅ ᓈᒻᒪᒋᔭᐅᓚᐅᕐᒪᑦ ᒥᓂᔅᑕᒧᑦ ᐊᒻᒪᓗ ᓇᓐᓄᒍᑏᑦ ᑐᓂᐅᖅᖃᐅᑕᐅᓂᖏᑦ ᓄᓇᓕᖕᓄᑦ. ᐅᖃᖅᑲᓂ ᐱᔾᔪᑎᖓᓂᑦ ᑲᑎᒪᑎᑦᑎᓂᐅᑉ ᑐᓂᓯᓗᑎᒃ ᓇᐃᓈᖅᓯᒪᔪᓂᒃ ᖃᓄᐃᙱᓂᐅᔪᓂᒃ ᖃᖓᑕᓱᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔾᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓚᓕᐊᔪᓕᕆᔾᑯᑦ ᐱᓕᕆᐊᖑᓐᓂᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐃᓱᒪᒋᔭᖑᓐᓂᒃ ᑐᓴᕐᓂᐊᕐᓗᑎᒃ.

ᖃᖓᑕᓱᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᑦ ᐊᔾᔾᑦ

- **ᓅᒃ**: ᖃᐅᔨᒋᐊᓚᐅᖅᐱᓯᐅᒃ ᓂᕆᖃᑦᑕᖅᑕᖏᑦ ᓇᓄᐃᑦ ᐊᕐᕕᐊᑦ ᖃᓂᑖᓂᑦ?
- **ᐊᕙᓐ**: ᐱᓕᕆᓂᖃᓚᐅᖅᑐᑦ ᒫᓂᑑᐸᒥᑦ. ᐸᐅᙱᖃᑦᑎᐊᖅᑐᑦ, ᓇᓄᕐᓂᑦ ᑕᑯᓯᒪᔪᖓ ᑐᖁᑦᑎᔪᓂᒃ ᓇᑎᕐᓂᑦ, ᐊᒻᒪᓗ ᓇᓄᐃᑦ ᑐᒃᑐᕕᓂᕐᓃᑦᑐᑦ.
- **ᓅᒃ**: ᖃᓄᖅ ᐊᖏᑎᒋᔪᒥᒃ ᖃᐅᔨᓴᐃᓂᕙᓯᖕᓄᐊᖅᓯᖃᑦᑕᖅᐸᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᓂᑦ?
- **ᐊᕙᓐ**: ᖃᑉᐳᓈᓂᑦ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓯᒪᑐᐃᓐᓇᖅᑐᖓ, ᑭᓯᐊᓂ ᐃᓕᑦᑎᕚᓪᓕᐊᔪᖓ ᐃᓚᓕᐅᑎᓂᐊᕐᓗᒋᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓃᑦ.

ᖃᖓᑕᓱᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᔪᑦ/ᐱᓕᕆᓃᑦ

- **ᑭᓷᑕᓐ** ᖃᐅᔮᓚᐅᖅᑐᖅ ᖃᐸᓯᐊᕐᔪᖕᓂᑦ ᐃᓚᐅᕐᑲᑕᐅᔪᑦ ᐊᑎᖏᑦ ᑖᒪᖅᑕᐅᓯᒪᔪᓂᑦ ᐅᓂᑉᑳᒥᑦ. ᐊᓕᓴ ᒪᒥᐊᑦᑐᖅ ᐊᒻᒪᓗ ᐅᖃᖅᑲᓂ ᖃᐅᔨᒋᐊᕐᓂᐊᖅᑕᖏᑦ ᐋᖅᑭᒃᑐᕆᓪᓗ.

- **ᑯᓇᒃ**: ᖃᓄᖅ ᓄᓇᖅᐸᓯᒃᑎᕆᔪᒧᑦ ᓈᓴᐃᖅᑲᑦᑕᖅᐱᓯ?
- **ᐊᓕᓴ**: ᖃᐅᔨᒋᐊᒃᑲᓐᓂᕐᓂᐊᖅᐳᖕᒐ ᐊᒻᒪᓗ ᖃᐅᔨᑎᓪᓗᓯ.
- **ᑯᓇᒃ**: ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᑕᓇᐅᕐᒦᖏᓐᓇᕐᓚᖃᑦᑕᖅᑐᑦ ᐊᐅᔭᓗᒃᑖᒥᑦᑕᐅᖅ.
- **ᐊᓕᓴ**: ᐋ, ᑕᐃᒪᓐᓇ ᑐᓴᓚᐅᖅᓯᒪᔪᒍᑦ. ᐊᔪᕐᓇᖃᑦᑕᖅᑐᖅ ᓈᓴᐅᑉᐳᓂ ᑕᓇᐅᕐᒥᑦ ᐊᑦᑕᕐᓇᖅᑑᖕᒪᑦ ᖁᓕᒥᒨᓕᒃᑯᑦ, ᑕᒪᓐᓇ ᐃᓱᒪᒋᔭᕗᑦ ᓯᕗᓂᖅᓴᒥᑦ ᓈᓴᐃᓐᓄᑦ ᒥᖅ ᑳᒡᐳᑦ ᐱᓕᕆᐊᕆᔭᖓ ᐆᒃᑑᑕᐅᓂᐊᕐᓗᓂ ᐃᓐᓄᒃᑕᐅᓯᒪᙱᑦᑐᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᐃᓂᖅ. ᐃᓱᒪᔪᒍᑦ ᐃᓐᓄᒃᑕᐅᓯᒪᙱᑦᑐᑦ ᖃᖓᑕᓲᑦ ᐊᔪᙱᓐᓂᐊᕐᒪᑕ ᓈᓴᐃᓂᐊᕐᓗᑎᒃ, ᐃᓐᓄᒃᑕᐅᓯᒪᙱᑦᑐᓂᑦ ᖃᖓᑕᓲᓂᒃ ᐃᒪᕐᒨᕆᐅᑎᑦᑎᔪᖕᓇᖅᑐᒍᑦ ᓈᓴᐃᓂᐊᕐᓗᑕ ᐃᒫᓃᑦᑐᓂᑦ.

- **ᑭᓷᑕᓐ**: ᓇᓄᐃᑦ ᐊᔾᔨᒌᙱᓐᓂᖅᑲᖅᐸᑦ ᐊᑯᓐᓂᖏᓐᓂᑦ ᑕᓯᐅᔭᕐᒥᑦ, ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸ?
- **ᐊᕙᓐ**: ᐊᔾᔨᒌᙱᓐᓂᖅᑕᕐᑳᕐᔪᒃᑐᖅ, ᑭᓯᐊᓂ ᐃᓗᐃᑦᑐᒥᐅᕆᑦᑐᖅ ᐊᑯᓐᓂᖏᓐᓂᑦ ᒪᕐᕈᒃ. ᑕᑯᔭᕗᑦ ᐊᓈᓇᐅᔪᖅ, ᐊᕐᓇᓪᓗᖅ ᐊᑎᖅᑕᖅ, ᐃᓚᒌᖕᒪᑎᒃ. ᐊᕐᓇᓪᓗᐃᑦ ᐅᑎᕋᔪᒃᑐᑦ ᐊᓈᓇᖏᑦᑕ ᐊᑎᖅᑕᖅᑖᕐᕕᒋᓚᐅᖅᑕᖏᓐᓄᑦ ᓯᑎᖏᓐᓄᑦ – ᑕᐃᑯᖓ ᐅᑎᖃᑦᑕᖅᑐᑦ, ᐊᖑᓴᓪᓗᐃᓪᓚ ᓇᒨᖓᐅᑲᑕᖃᑦᑕᕐᓂᖅᓴᐅᔪᑦ. ᑲᔪᓯᓂᖃᖅᑐᖅ ᐃᓚᒌᖕᓂᖏᓐᓂᑦ ᓄᓇᐅᔪᒥᑦ. ᐊᕕᒃᓯᒪᓪᓚᑦᑖᕐᓂᖅᑕᖃᙱᑦᑐᖅ ᐊᔾᔨᒌᙱᓂᖏᓐᓄᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᐅᓚᓂᕆᔭᖏᓐᓄᑦ.

- **ᓅᒃ**: ᐊᑐᖅᑕᐅᖃᑦᑕᖅᑐᑦ, ᐱᑕᖃᙱᑦᑐᖃᐅᖅᑎᑦᑎᔪᑦ. ᐅᖓᓯᒍᐊᖏᑦᑐᑎᑦᓄᓇᖅᐸᓯᒃᑐᒥᑦ. ᖃᐅᔨᒪᔭᐅᔪᑦ 200 ᒪᐃᓕᓂᑦ ᐅᖓᓯᒃᑎᕆᔪᑦ ᓯᒡᔭᒥᑦ. ᐱᖃᑦᑕᕆᒃᑯᐸᓯ 5-ᓂᒃ 10%-ᖏᓐᓂᓪᓘᓐᓃᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓇᓄᕐᓂᑦ, ᐊᒥᓱᙳᕆᐊᖅᓯᓂᐅᔪᖕᓇᖅᑐᖅ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑉᑎᖕᓄᑦ. ᓲᕐᓗ ᐱᔪᖕᓇᖅᑎᑦᑎᔨᐅᔪᑦ, ᐃᓅᕐᓇᓕᖅᑐᑦ ᐊᒥᓱᙳᕆᐊᖅᓯᖃᑦᑕᕆᓐᓂᖏᓐᓂᒃ ᓱᓇᑐᐃᓐᓇᓂᒃ ᐱᔭᐅᔪᖕᓇᖅᑐᓂᒃ, ᐅᒥᖕᒪᐅᒐᓗᐊᖅᐸᑕ ᐅᕝᕙᓘᓐᓃᑦ ᓇᓄᐃᑦ, ᐱᒋᐊᕆᐊᒃᑲᐅᑎᕆᔪᖕᓇᖅᓗᑎᒃ ᐊᒥᓲᔪᖕᓃᖅᓯᕆᐊᕐᓂᐊᕌᖕᒐᑕ ᐱᑕᖅᑕᐅᔪᖕᓇᖅᑐᓂᒃ. ᑕᐃᒪᓐᓇᒧᑦ ᐱᒻᒪᕆᐅᔪᖅ ᖃᖕᒐᒃᑯᑦ ᐅᑭᐅᑉ ᐃᓗᐊᓂᑦ ᓈᓴᐃᖃᑦᑕᕐᓗᑎᒃ, ᖃᐅᔨᔪᖕᓇᖅᑕᓯ ᐊᒥᓱᙳᖅᐹᓪᓕᖅᓯᒪᓂᖏᑦ ᐃᓄᖕᒧᑦ ᓴᓇᔭᐅᔪᒃᑯᑦ ᑭᒡᓕᓕᐅᖅᓯᒪᔪᒧᑦ. ᓇᓗᓇᐃᖅᓯᓪᓚᑦᑖᖅᓯᓂᖅ ᑕᐃᒪᓐᓇᐅᖓᑦ ᐊᔪᕐᓇᖅᑐᖅ, ᑭᓯᐊᓂ ᐃᓱᒪᒋᔭᐅᔪᖕᓇᖅᑐᖅ ᑕᑯᔭᑉᑎᒍᑦ, ᐱᓗᐊᖅᑐᒥᑦ ᐊᕐᕕᐊᓂᑦ, ᓄᓇᖅᑕᖃᐅᕐᓂᖅᓴᐅᔪᖅ. ᐃᒪᖅᖄ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᐊᒥᓲᔪᓐᖅᑐᑦ ᐊᒻᒪᓗ ᐊᑐᖅᑐᐊᕐᓚᒃᑐᖅᖃᐃᑦ ᐃᓚᖓᓐᓂᑦ ᐱᖓᖕᓇᖅᐸᓯᐊᓃᑦᑐᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᕐᓂᑦ. ᑕᒡᕙᓂ ᑕᒡᕙ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐱᒻᒪᕆᙳᑦᑎᐊᖅᑐᑦ ᐱᕆᐊᓷᕝᕕᐅᔪᒥᑦ ᑕᒪᐃᓐᓂᓗᒃᑖᖅ ᐱᓕᕆᐊᖑᔪᒥᑦ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔩᔪᐊᒃᑯᓐᓂᐅᖏᑦᑐᖅ. ᐃᒻᒪᖅᖄ ᑕᐃᑲᓂ ᓈᓴᐃᖅᖃᐅᑕᐅᔭᕆᐊᖅᑲᖅᑐᓯ, ᑕᑯᔭᓯ ᐊᔾᔨᒌᙱᓐᓂᖏᓐᓂᑦ 1990-ᒥᑦ ᐊᒻᒪᓗ 2023-ᒥᑦ, ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ. ᑕᒪᕐᒥᓗᒃᑖᑦᑎᐊᖅ ᐊᕐᕕᐊᕐᒥᐅᑕᐃᑦ ᐅᖃᕐᓂᐊᖅᑐᑦ ᓇᓄᖅᑕᖃᐅᕐᓂᖅᓴᐅᓕᖅᑐᖅ. ᓈᓴᐃᓂᓯ ᑕᑯᒃᓴᐅᑎᑦᑎᔪᑦ ᓇᓄᖅᑕᖃᐅᖏᓐᓂᖅᓴᐅᖕᒪᑦ.
- **Hᐃᐅᕆ**: ᑲᑎᒪᔨᐅᖃᑕᐅᓕᖅᑐᖓ, ᐱᒋᐊᓖᕐᓂᖏᓐᓂᑦ ᐅᑭᐅᖅ 2000-ᓂᑦ. ᑐᓵᕙᓚᐅᖅᑕᕋ ᐊᑖᑕᒐ ᐊᓯᖏᑦᑐ ᐃᓐᓇᕆᔭᐅᔪᑦ ᐅᖃᓪᓚᒃᑎᓪᓗᒋᑦ ᓇᓄᐃᑦ ᓈᓴᖅᑕᐅᔪᓂᑦ, ᐅᑭᐅᑦ 80-ᖏᓐᓂᓪᓘᓐᓃᑦ ᐊᒻᒪᓗ 90-ᖏᓐᓂᑦ, ᐅᖃᐃᓐᓇᖃᑦᑕᓚᐅᖅᑐᑦ ᐊᑐᖅᑕᐅᔪᖅ ᓈᓴᐃᓂᕐᒧᑦ ᓇᓚᐅᑎᓯᒪᙱᑦᑎᐊᖅᑐᒡᔫᖅ (ᓇᓚᐅᑦᑖᖅᓯᒪᔪᑦ). ᓇᓗᓇᙱᑦᑐᖅ ᐅᐸᓗᒥᐅᓕᖅᑐᖅ ᐊᒥᓲᓂᖅᓴᙳᖅᐸᓪᓕᐊᑐᐃᓐᓇᖅᑐᓂᒃ ᓇᓄᕐᓂᑦ ᑕᑯᖃᑦᑕᖅᑐᒧᑦ ᓄᓇᓕᐸᑕ ᖃᓂᑖᓃᑦᑐᓂᑦ ᐅᔭᐅᑎᓪᓗᒍ. ᑕᐃᒪᓐᓇ ᐃᓅᐃᓗᑕᓂᒃ ᐱᑕᖅᐸᓚᐅᖅᓯᒪᙱᑦᑐᒍᑦ. ᐅᐸᓗᒥ, ᐊᒥᓱᑦ ᐃᓄᐃᑦ ᑐᐱᕐᒥᔾᔮᖏᓕᖅᑐᑦ ᐊᒥᓱᙳᓗᐊᕐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ. ᓄᓇᓕᕗᓪᓗᓐᓃᑦ, ᑕᒪᐃᓐᓇ ᐃᓅᐃᓗᑕᖅᑕᖅᐸᓪᓚᐅᖅᓯᒪᙱᑦᑐᒍᑦ. ᐅᐸᓗᒥ ᖃᐅᔨᔭᐅᖃᑦᑕᕐᓂᖅᓴᐅᓕᖅᑐᑦ ᓇᓄᐃᑦ. ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓ ᐱᔾᔪᑕᐅᔪᖅ? ᖃᐅᔨᒪᙱᑦᑐᖓ. ᐃᒻᒪᖄᖅ

ᐱᔾᔪᑕᐅᓗᐊᖅᑐᖅ. ᐊᒥᓯᑦ ᓇᓄᐃᑦ ᐅᑎᖃᑦᑕᙱᓕᖅᑐᖅ ᐊᐅᔭᖅᓯᐅᕝᕕᒋᒐᔅᑕᕐᕆᓐᓄᑦ. ᐊᒥᓲᓂᖅᓴᐃᑦ ᑕᑯᔭᐅᖃᑦᑕᕐᓂᖅᓴᐅᓕᖅᑐᑦ ᖃᓂᑦᑐᒥᑦ.

- ᓈᓴᐃᓂᖅ ᓈᒻᒫᓂᙱᑦᑐᖅ. ᓇᓚᐅᑦᑎᓯᒪᔫᓂᕋᕐᓂᖅ ᐊᔪᖅᑕᕋ. ᖃᓂᙱᑦᑎᐊᖅᑐᕐᓘᓐᓃᑦ. ᑐᓵᖃᑦᑕᖅᓯᒪᓕᖅᑐᖓ ᐊᒥᓱᓄᑦ ᐅᑭᐅᓄᑦ. ᖃᐅᔨᒪᐃᓐᓇᖅᑐᑦ ᓄᓇᖅᐸᓯᒃᑑᒦᖃᑦᑕᖅᑐᑦ. ᐊᒥᓲᕐᒋᑦᑑᕐᒋᑦᑐᑦ, ᑭᓯᐊᓂ ᐊᒥᓯᐊᓗᐃᑦ ᐊᕐᒋᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ. ᓂᕈᐊᑏᑦ ᑭᒡᓕᖅᑲᙱᑦᑐᑦ. ᐃᓄᐃᑦ ᑭᒡᓕᕐᒋᑦ ᓇᓚᐅᑦᑎᓯᒪᙱᑦᑎᐊᖅᑐᑦ.
- **ᑖᓇ:** ᐊᐅᓚᑦᑎᔪᓐᓇᕐᓂᖅᑲᓇᐊᖅᑲᖅᑐᑎᑐᑦ ᑭᒡᓚᐅᔪᑦ.
- ᐱᒋᐊᖅᑐᑎᑦ ᐃᖅᑲᐅᒪᓂᓇᔭᕐᓕᓂᑦ ᐃᓅᔪᕐᓃᓚᐅᖅᑑᑉ ᐱᓕᕆᖅᑎᒋᓚᐅᖅᑕᕕᑦ. ᐃᓅᔪᕐᓃᖅᑐᖅᑲᐅᖅᓯᒪᔪᒍᑦ. ᐅᑎᖅᑎᑦᑎᓂᖅᑲᖅᑐᖅ. ᑐᖅᑯᔪᑦ, ᓇᓄᖅᓄᑦ ᑐᖅᑯᑕᐅᔪᑦ. ᐊᒻᒪᓗ ᐃᓅᔪᕐᓂᖅᖃᖅᓯᒪᔪᑦ ᐊᑐᖅᓯᒪᔪᓂᒃ. ᖃᐅᔨᒪᔪᕐᓗ ᐅᖃᐅᓯᓇᔭᐃᑦ ᐃᓄᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᖅᑲᙳᕐᒋᑦᑐᕐᒦᑦᑕᓇᐊᖅᑲᕐᓂᕐᒋᓐᓂᒃ, ᑭᓯᐊᓂ ᐃᓚᐅᑎᓪᓗᒋᑦᑕᐅᖅ ᑐᖅᑯᑕᐅᔪᑦ, ᐃᓯᒪᔪᕐᓗ. ᐊᑕᐅᓯᖅ ᐃᓅᓯᖅ ᐊᒥᓲᓗᐊᖅᑐᖅ.

- **ᓵᓪ:** ᑐᕌᓴᓇᔭᖕᓴᓐᓄᑦ ᓴᕙᒪᒃᑯᑦ, ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓕᐊᓂᕐᒥᑦ ᐱᖃᓕᖅᑐᓯ, ᓂᕈᐊᑎᓄᐊᕈᑎᒋᐸᓗᒍ. ᓱᓇᒥᒃ ᑎᑭᐅᑎᓂᖅᑲᕈᒪᕕᑦ? ᓱᓇᒧᑦ ᓴᕙᒪᒃᑯᑦ ᑎᑭᐅᑎᓂᖅᑲᕈᒪᕙᑦ?
- **ᐊᕙᓐ:** ᐊᐱᖅᑯᑎᒥᑦ ᐊᐱᕆᔭᐅᒐᔪᒃᑐᒍᑦ ᐊᒻᒪᓗ ᐃᓯᒪᒋᖃᑦᑕᐃᓐᓇᖅᑲᖅᑕᕋ ᐱᓕᕆᐊᓪᓃᑎᓪᓗᒍ. ᐊᑕᐅᓯᖅ ᐱᓗᐊᖅᑐᖅ ᐃᓯᒪᒋᔭᕋ ᑭᑐᓗᒃᑖᑦ ᑕᒡᕙᓃᖃᑕᐅᔪᑦ ᐃᒻᒪᖅᑳ ᐊᕐᒋᖃᑕᐅᔪᖕᓇᖅᑐᑦ ᑕᒪᑉᑕ ᓇᓄᕐᓂᑦ ᑕᑯᖃᑦᑕᕈᒪᒐᑉᑕ ᓯᕗᓂᒃᓴᒥᑦ. ᓇᓄᖅᑕᖃᐃᓐᓇᖅᑯᐸᓗᒃᑖᖅᑐᒍᑦ, 10, 15, 20, 60, 80, Hᐊᓐᓇᐃᑦ ᐅᑭᐅᓂᑦ ᐅᐸᓗᒥᒥᑦ ᐊᑯᓂᐅᔪᒧᑦ ᐊᒥᓱᓄᑦ ᐅᑭᐅᓄᑦ.
- **ᓵᓪ:** ᓇᓗᓇᙱᓕᖅ ᐅᖅᑲᑦᑕᓇᒃᓴᖅᑲᑎᖅᑲᖅᓯᒪᒪᕕᑦ ᐃᓄᖕᓂᑦ, ᑐᒃᑐᓄᑦ ᐆᒪᔪᓕᕆᔨᓂᑦ, ᐃᖅᑲᓗᓕᕆᔨᓂᑦ, ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ. ᐊᑕᐅᓯᕐᒥᒃ ᐆᒪᔪᒥᒃ ᐊᑦᑎᖅᓯᔪᖕᓇᖅᐱᑦ ᐃᓄᖕᓄᑦ ᒪᖅᑲᐃᑕᐅᓲᒥᒃ ᓄᖑᔅᓱᑎᒃ ᐱᑕᖃᕈᖕᓃᖅᑐᓂᒃ?
- **ᐊᕙᓐ:** ᐋᒃᑲ.
- **ᓵᓪ:** ᓲᓕᓇᕙᑕᓕ ᑕᒡᕙ ᐋᓪᓕᕋᐅᔭᕐᒥᑦ? ᐱᕝᑕᑉᑎᖕᓂᑦ ᐱᑎᑎᒍᑦ. ᐃᓅᓯᖃᖅᑎᑎᒍᑦ ᐃᓅᓂᓇᖃᑦᑕᖅᑕᑉᑎᖕᓂᑦ. ᐃᓱᐃᓗᑕᖅᑲᑕᖅᑲᓕᓲᖅ. ᓲᓕᔪᖅ, ᖃᓄᖅ ᐃᑲᔪᕐᓂᐊᖅᐸᑦ ᑲᑎᒪᑎᑦᑎᓃᑦ? ᐃᓅᓯᖃᖅᑎᑎᒍᑦ.
- **ᐊᕙᓐ:** ᐊᕙᓗᔭᐅᓯᒪᔪᒍᑦ ᐋᖅᑭᐅᒪᔪᒧᑦ ᓇᓚᐅᑉᐸ?
- **ᓵᓪ:** ᐋᖅᑭᐅᒪᔭᓯ, ᐋᖅᑭᐅᒪᔫᕐᒋᑦᑐᖅ.
- **ᐊᕙᓐ:** ᐃᒻᒪᖄ, ᓄᓇᑖᖅᓯᒪᓂᖅᒧᑦ ᐊᕐᒋᖃᑎᒌᒍᑎ.
- **ᓵᓪ:** ᑖᒻᓇᓘᓐᓃᑦ ᐃᓚᑉᓯᕐᓄᑦ ᐋᖅᑭᐅᒪᔫᕐᒪᑦ. ᐃᓄᖕᒧᑦ ᑐᖅᓯᕐᓗᔪᒥᒃ ᐱᑕᖃᙱᑦᑐᖅ.
- **ᑖᓇ:** ᑕᐃᒪᐅᔪᑦ, ᑭᓯᐊᓂ ᓯᖁᑉᑎᖅᓯᒪᔪᐃᓐᓇᐃᑦ ᐋᖅᑭᐅᒪᔪᑦ, ᓇᓃᓐᓂᐅᓗᐊᖅᑐᒥᑦ, ᐱᓕᕆᐊᖑᑦᑎᐊᕆᐊᖅᑐᑦ.
- **ᓵᓪ:** ᓯᓚᕗᖅᓕᕆᓚᐅᖅᑐᖅᓚ ᐅᓐᓄᒃ ᐊᒻᒪᓗ ᐃᓯᒪᓚᐅᖅᑐᖅᓚ ᓲᖅ ᒪᕐᕉᖕᓂᒃ ᐃᑲᕐᕋᖕᓂᒃ ᐊᑐᕆᐊᖃᕐᓂᐊᖅᐳᖅᓚ? ᓱᓇᒥᒃ ᑕᑯᔭᒪᒐᖅᓚ? ᓱᓇᒥᒃ ᐃᓄᐃᑦ ᑕᑯᔭᒪᕙᑦ? ᓱᓇᒧᑦ ᑕᐃᒪ ᑎᑭᐅᑎᓂᐊᖅᐱᑕ?
- **ᐊᕙᓐ:** ᑕᐃᒪᑐᖅ ᐃᓚᖕᒋᓐᓂᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᑐᓂᓯᕗᖃᕐᓂᐊᖅᐳᖓ ᐊᒻᒪᓗ ᐅᖃᓪᓚᒍᖃᕐᓗᖓ.
- **ᓵᓪ/ᑖᓇ:** ᑐᓂᓯᖃᑦᑕᖅᓯᒪᓕᖅᑐᒍᑦ ᑐᑭᓯᐊᒪᔾᔭᑎᓂᒃ 30-ᓄᑦ ᐅᑭᐅᓄᑦ, 40-ᓄᑦ ᐅᑭᐅᓄᑦ, 50 ᐅᑭᐅᓄᑦ ᐊᒻᒪᓗ ᑕᑯᓚᐅᖅᓯᒪᙱᑦᑐᒍᑦ ᑖᑉᑯᓇᓃᑦᑐᓂᑦ (ᑕᑯᔭᒃᓴᐅᑎᓂᑦ/ᐅᓂᑉᑳᓂᑦ).
- **ᐊᕙᓐ:** ᐃᒻᒪᖄ ᑖᓇ ᐱᒻᒪᕆᐅᔪᖅ ᑲᑎᒪᑎᑦᑎᓂᕐᒥᙳᕝᓯᓂᐊᖅᑐᖅ ᐃᓚᑉᓯ ᐊᓚᐊᕆᐅᖅᑐᕐᒦᓇᑉᓯ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᕐᒥᒃ ᐃᓯᒪᒋᔭᐅᓯᒪᓂᖅᓴᐅᕙᓪᓕᐊᑎᒋᒪᑕ ᐊᒻᒪᓗ ᐃᓚᑕᐅᑎᔭᐅᑲᓐᓂᖅᓴᐅᓇᐊᖅᑲᒪᑕ ᐊᐅᓚᑦᑎᓂᖅᓯᐅᑦ ᐋᖅᑭᐅᒪᔪᒧᑦ ᐊᒻᒪᓗ ᐱᔾᔪᑕᐅᔪᒧᑦ ᓈᓴᐃᓂᖅᑕᖃᖅᑎᑕᐅᓇᐊᖅᑲᒪᑕ ᑲᑎᖓᓂᐊᖅᓯᒪᔪᑦ ᐊᒻᒪᓗ ᐋᖅᑭᒃᓯᒪᓕᕐᓗᑎᒃ ᐃᓚᐅᑎᓐᓂᐊᖅᓱᒋᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᓇᓄᐃᑦ ᒥᒃᓵᓄᑦ ᐊᒻᒪᓗ ᐱᕙᓪᓕᐊᔪᓕᕆᓂᕐᒧᑦ ᑕᕐᓂᓗᐊᓗᐃᓇᕆᔭᐅᕙᒃᑐᓂ. ᑕᒪᓐᓇ ᐱᔪᓐᓇᓂᖅᓴᐅᓇᐊᖅᑐᖅ ᓯᕗᓂᖓᒍᑦ.
- **ᓵᓪ:** ᐃᓪᒋᐅᑎᑦᑕᐃᓐᓇᖃᕐᒥᒃ ᐃᓪᒋᖅᓯᒪᔭᕆᓕᑦ.
- **ᑖᓇ:** ᐱᑦᑕᐅᔪᒥᒃ ᓄᓇᒥᑦ ᐱᑎᑦᑎᓯᒪᙱᑦᑐᖅ ᐅᕙᑎᓐᓄᑦ. ᖃᓄᐃᓐᓇᖅᑐᖅ ᐋᖅᑭᒃᑲᓯᒪᓗᐊᖅᓂᖅᓚ. ᖁᕙᐊᓇᙱᑦᑐᒥᒃ ᐱᑕᖃᖅᑎᑦᑎᔪᖅ.
- **ᐊᕙᓐ:** ᑕᒪᓗᒃᑖᖅ ᐊᐅᓚᑦᑎᓂᐅᔪᖅ?
- **ᑖᓇ:** ᐋ, ᓈᓴᐃᓂᖅ, ᓈᓴᖃᑦᑕᙳᕐᑕᑦᑐᖅ ᓂᕐᔪᑎᖅᑯᑎᙳᖅ. Hᐃᐅᓇ ᐅᖃᖅᑲᐅᔪᖅ, ᐊᑖᑕᖓᓗ ᓯᕗᓕᖅᑎᐅᑎᐊᕙᓚᐅᖅᑐᖅ, ᓴᕕᐅᑉ ᐊᑖᑕᖓᓗ. ᐱᑕᖃᖅᑐᓐᓃᖅᑐᑦ, ᐊᒻᒪᓗ ᑕᐃᕙᓂᖅᑐᒍᑦ ᑐᓂᓯᕗᖃᖅᑐᑦ ᖃᐅᔨᒪᓂᖅᑕᖃᕐᓂᖅ. ᐱᓗᐊᖅᑐᒥᑦ ᐅᖃᖅᑲᐅᒪᔭᐅᑎᑦ, ᐊᔨᐅᖅᓯᕗᓐᓂᖃᑦ ᐊᒻᒪᓗ ᐅᐸᓗᒥᒥᑦ ᖁᓕᑦ ᐅᑭᐅᑦ ᐊᓂᒍᖃᖅᑎᑦᑎᓗᑦ, ᓂᕈᓕᕙᑉᐸᓂᐊᖃᖅᑎᓪᓗᒍ ᐊᔭᐱᑎᐊᕆᓂᐊᕐᓖᑎᑦ ᓱᓕ? ᑕᐃᒪᓐᓇ ᐊᐅᓚᑦᑕᖅᑕᙳᕐᑕᑦᑐᑦ ᓂᕐᔪᑎᖅᑯᑎᓐᓄᑦ ᐊᒻᒪᓗ ᓯᓚᓄᑦ ᐊᒻᒪᓗ ᐊᕙᑎᖅᑯᑦ. ᐊᔪᓇᖅᑕᖅᑕᑦᑐᒍᑦ ᑕᒪᕐᒥᒃ ᐊᓯᐊᒍᑦᑐᓄᑦ, ᐊᔪᓇᐅᖅᑐᓇ ᓯᔾᔪᐃᓐᓇᖅᑐᒍᑦ. ᐊᓯᐊᒍᔾᔭᖅᑐᖅ ᓈᓴᖃᑕᐅᓪᓗᑕ. ᐱᑕᖃᖅᑐᓐᓃᖅᐸᑦ, ᐃᓚᐅᓂᐊᖅᐳᒍᑦ ᖁᕝᕙᐃᓕᓂᐊᖅᓯᒪᔪᑦ ᑭᖑᓪᓕᖅᓯᒪᔪᖕᓂᒃ ᐅᐸᓗᓯᒪ. ᐱᑦᑕᐅᔪᓐᓂᐊᕐᓂᖅᓴ ᓂᕐᔪᑎᓂᒃ ᓯᕗᓂᐊᓂ.

- **ᔪᐋᓐ:** ᓱᓇᒥᒃ ᐳᓴᓐᑎᐅᕙᑦ ᓇᓐᓄᒃᑕᐅᔪᑦ ᓄᓇᕗᒻᒥᓗᒃᑖᖅ?
- **ᐊᓕᓴ:** ᓖᓐᓇ ᖃᐅᔨᒪᙱᑦᑐᖓ ᖃᑉᓯᐅᖕᒪᖔᑕ. ᑲᑎᓗᒃᑖᖅᓗᒥᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐋᖅᑭᒃᑕᐅᓯᒪᕙᓚᐅᖅᑐᑦ 4.5%-ᖑᓂᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᒥᑦ ᐊᒥᓲᑎᒋᓂᖕᒋᓐᓄᑦ. ᑭᖑᓪᓕᖅᐹᖅ ᐃᓯᒪᓕᐊᖑᓚᐅᖅᑐᖅ ᐱᖕᖑᓇᖕᖑᓄᑦ ᑕᓯᐅᔭᖅᔪᐊᖅᒧᑦ, ᐊᒥᓯᖕᒍᓇᐊᖅᑕᐅᒃᓪ 38-ᓄᑦ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐊᓯᐊᒍᓚᐅᖅᑕᖏᑦ 38-ᒧᑦ 38-ᖑᔪ 4.5%-ᖑᓚᐅᖅᓯᒪ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᓂᑦ.

- **ᔪᐋᓐ**: ᓱᓕᕖᑦ? ᑐᓵᖅᑕᕋᒪ 3%-ᖑᓂᑦ.
- **ᓵᒻ**: 3%-ᖑᓂᐅᕙᓚᐅᖅᑐᖅ. 2%-ᖑᓂᐅᕙᓚᐅᖅᑰᖅᑐᖅᓘᓐᓃᑦ, ᐊᑦᑎᒃᑐᒻᒪᕆᐊᓘᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᐊᖏᖅᑕᐅᓗᑕ ᑭᓯᐊᓂᐅᓚᐅᖅᑐᖅ. ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ ᑎᑭᐅᑎᖕᒪᑦ ᑐᓵᖅᑕᐅᔪᒧᑦ, ᐳᓴᓐᑎᖑ ᐊᒥᓱᙳᓇᐊᕐᓂᐊᖅᓱᓂ. ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐊᖏᒡᓕᒋᐊᓇᐊᖅᑲᓚᐅᖅᑐᖅ, ᑭᓯᐊᓂ ᐊᖏᒡᓕᒋᐊᓚᐅᖏᑦᑐᖅ ᒪᕙᒃᑯᑦ ᐊᓯᐊᓄᐊᖏᓐᓇᖅᑲᑦᑕᖅᒪᔾᔪᒃ ᖃᓄᐃᖏᑦᑐᑦ ᐅᒥᓲᑎᒋᓂᖅᓄᑦ ᓈᓴᐅᑎ. ᐃᓛᓐᓂᒃᑯᑦ 2000-ᖑᕙᒃᓱᓂ, 3000, 5000-ᖑᕙᒃᓱᓂ, ᓇᓕᐊᑐᐃᓐᓇᖅ. ᐊᐅᓚᓂᖅᑲᖅᑐᖅ ᑐᓵᖅᑕᐅᔪᖅ.

- **ᔪᐋᓐ** ᖃᐅᔨᑎᑦᑎᔪᖅ ᐃᓱᒫᓘᑎᒋᔭᖑᓂᒃ ᐃᓄᐃᑦ ᐊᑦᑕᖅᓇᖅᑐᒦᓐᓂᖏᑦ ᓄᓇᒦᑎᓪᓗᒋᑦ.
- **ᓵᒻ**: ᑐᐱᖅᓯᒪᓗᑕ ᑕᖕᒫᖅᑲᑦᑕᕈᖕᓇᖅᐸᓚᐅᖅᑐᒍᑦ. ᕿᖕᒥᖅᑲᓇᐊᖅᑲᖅᓇᖅᐸᓚᐅᖏᑦᑐᖅ. ᐱᑭᐅᑕᖅᐸᓚᐅᖅᑐᒍᑦ ᓄᑕᖃᐅᑉᓗ ᐊᖏᖅᑲᓇᔭᑉᑎᖕᓄᑦ ᐅᖑᓯᒃᑐᒥᑦ ᓯᑉᔭᓂᑦ. ᑕᐃᒪᓐᓇᐃᓐᓂᖅ ᐊᔪᖅᓇᖅᓯᔪᖅ.
- **ᑯᓇᔅ**: ᒫᓐᓇᓘᓐᓃᑦ, ᐊᐅᓪᓚᕈᑉᑕ, ᐅᒥᐊᕐᒥᑦ ᓯᓂᖅᑲᑦᑕᖅᑐᒍᑦ ᑐᐱᕐᒦᑦᑕᕌᖑᓚᑉᑕ, ᒪᕐᕈᐃᖅᑕᖅᑐᑕ ᑐᐸᖅᑲᑦᑕᓚᐅᖅᑐᒍᑦ ᐊᒻᒪᓗ ᒪᕐᕉᖕᓂᑦ ᐱᖓᓱᓂᑦᓘᓐᓃᑦ ᓇᓄᖅᑕᖅᑲᖅᓗᓂ ᑐᐱᐅᑉ ᖃᓂᑖᓂᑦ, ᑕᐃᒪᓐᓇᓗᑦ ᑐᐱᕐᒥᑦ ᐊᑐᖅᑲᑦᑕᕈᖕᓇᖅᑐᒍᑦ.
- **ᑭᓛᑕᓐ**: ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐃᒪᓐᓇᐃᓴᕐᕕᖃᖅᑐᓂᒃ ᐃᓯᒪᒋᓪᓗᒋᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᓂᒃ ᐊᒥᓲᔪᖕᓃᖅᑎᑦᑎᓂᐊᕐᓂᐊᕐᓗᑎᒃ ᐊᒥᓲᑎᒋᓂᖏᑦ ᓇᓄᐃᑦ. ᐃᓯᒪᒋᔭᐅᓚᐅᖅᓯᒪᕙ ᑕᕝᕙᓂ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓄᑦ ᐱᖅᑯᓯᕐᓂᐅᔪᓄᑦ?
- **ᐊᓕᔅ**: ᐋᒃᑲ ᐃᓯᒪᒋᔭᐅᓚᐅᖏᑦᑐᖅ, ᑭᓯᐊᓂ ᑕᒪᓐᓇᖕᒌ ᐃᓚᒋᔭᐅᔪᖅ ᐅᖅᑲᐅᓯᐅᓯᖃᖅᓯᒪᑎᓂᐊᕐᓂᖅ? ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐃᓯᒪᓕᐅᕆᒪᒍᑎᒃ ᐊᐅᓚᑦᑎᓂᖅᒧᑦ ᑐᓵᕙᐅᔪᒥᒃ, ᑎᑎᖅᑯᓯᓇᔾᔪᓇᖅᑕᖁᑦ ᑎᑎᖅᑲᓂᑦ ᐊᒻᒪᓗ ᐃᓚᓕᐅᑎᓯᐅᔾᔪᓇᖅᑐᑦ. ᐆᒃᑑᑎᒋᓗᒍ, ᑕᐃᒪᓐᓇᐃᓚᐅᖅᑐᑦ Davis Strait-ᒥᑦ (ᑕᓇᐅᖅ ᐊᑯᓐᓂᖓᓃᑦᑐᖅ ᕿᑭᖅᑖᓗᒃ ᐊᒻᒪᓗ ᐊᑯᑭᑦᑑᑉ) ᓄᓇᖅᓄᑦ. ᐱᓕᕆᐊᙳᖅᑳᖅᑎᓐᓇᖓ, ᐃᓯᒪᓕᐅᕆᓂᖏᑦ ᓄᓈᑦ ᐅᖅᑲᓚᐅᖅᑐᑦ, ᐊᐅᓚᑦᑎᓂᖅᒧᑦ ᐱᔭᒃᓴᐅᑎᖏᑦ ᐊᒥᓲᔪᖕᓇᐊᖅᑐᑦ ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᐃᓚᓕᐅᑎᔭᐅᓚᐅᖅᑐᖅ ᑕᒪᓐᓇ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓄᑦ. ᓄᑕᐅᓛᖅ ᐃᓯᒪᓕᐊᖑᓚᐅᖅᑐᖅ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᓐᓄᑦ ᐊᒥᓱᙳᓇᐊᕐᓗᒍ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ 4-ᓂᒃ ᐃᓚᓯᕗᖅᐸᐅᓗᓂ. ᑕᑯᓯᐅᑎᑦᑐᒪᖅᑲᐅᔪᖅ ᐱᔭᕇᖅᑐᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐊᒥᓱᙳᓇᐊᖅᓯᖅᑲᑦᑕᖅᑐᑦ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ ᐃᓛᓐᓂᒃᑯᑦ. ᐊᒥᓱᙳᓇᐊᖅᓯᓚᐅᖅᑐᑦ ᐃᒪᖅ ᐴᑎᐊᒥᑦ (Gulf of Boothia), M'Clintock ᓯᐸᓗᒧᑦ, ᐊᒻᒪᓗ Davis Strait-ᒧᑦ.
- **ᑭᓛᑕᓐ**: ᓯᓚᓂᖅᒍᑦ ᐱᔭᕇᖅᑕᐅᕙ ᐅᖅᑲᖅᑎᖅᑲᖅᓂᖅᒍᑦ ᖃᐅᔨᓂᐊᖅᑐᓂ ᐊᐅᓚᑦᑎᔩᑦ ᐱᔭᒃᓴᐅᑎᖓᑦ?
- **ᐊᓕᔅ**: ᐋᒻᒪᐃ, ᐄᖑᔪᖅᑕᖅᑐᖅ, ᐃᓯᒪᔪᖅ. ᐃᓯᒪᔪᖅ ᒫᓇ ᐱᓗᒍ ᑭᓯᐊᓂ, ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᑕᖅᑲᖅᐸᑦ.
- **ᐊᔾᔨᐊ**: ᑕᐅᑐᒃᓯᒪᔾᔭᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᑭᔾᔪᐅᔾᔪᑦ ᐊᒻᒪᓗ ᐃᓯᒪᔪᖅ ᑭᑐᑦ ᖃᐅᔨᖅᑕᖅᑐᒪᖑᕕᒋᑦ, ᐊᒻᒪᓗ ᕿᓂᖅᓯᒪᐃᓐᓇᖅᑐᖅ ᐱᔾᔪᑎᐅᔪᓄᑦ ᐊᓯᖏᓐᓂᑦ ᐃᓂᐅᔪᓂᑦ. ᓇᓄᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᖕᒪᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᓐᓄᑦ ᓈᓚᒋᔭᐅᔭᕆᐊᖅᑲᖅᓱᑎᒃ ᐊᒻᒪᓗ ᓄᓇᕗᒻᒥ ᓴᐳᒻᒥᒃᑯᑦ ᓈᓚᒋᓪᓗᑎᒃᑕᖑᒐᓗᐊᖅᑕ, ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᐅᔪᑦ ᐅᖃᐅᓯᐅᓯᒪᓂᕐᓴᖑᒃᑯᑦ ᓄᑖᒥᒃ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᒃ ᐱᑕᖅᑲᕈᓐᓃᖕᒪᑦ ᖃᐅᔨᓴᐅᓂᐊᕐᓗᑎᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐃᓯᒪᓕᐅᕆᐊᖅᑐᖅᑕᖅᑐᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᓄᑦ. ᐅᖅᑲᓕᓚᐅᖅᑕᖅᓯᓂᑦ, ᑕᐃᒪᓐᓇᐃᓚᐅᖅᓯᒪᓕᖅᑐᖅ ᓇᓕᐊᑐᐃᓐᓇᖅᓂᑦ ᐱᓕᕆᐊᖑᔪᓂᑦ. ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐅᖅᑲᓗᐊᖅᑎᓪᓗᑎᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᑕᖅᑲᖅᑐᖅ, ᓄᓇᕗᒻᒥ ᓴᐳᒻᒥᒃᑯᑦ ᐃᓯᒪᓕᐊᓇᖏᑦᑕᖑᑦ. ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᑉ ᑎᑎᕋᖅᓯᒪᓂᖅᓯᖑ ᐃᓯᒪᓕᐅᕆᓂᕐᒥᐅᓯᓇᐊᖅᑕᖅᑐᖅ. ᓇᓄᖅᑕᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᒥᑦ, ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᖁᔭᓂᖅᑕᖅᑐᖅ, ᐱᔭᒃᓴᐅᑎᓂᐅᖅᑐᖅ, ᐅᖅᑲᖅᓯᒪᓂᖑ ᑕᐃᒪᓐᓇᓗᑦ ᐱᔾᔪᑎᖃᕐᓂᐊᖅᑕᖅᑐᖅ. ᐊᑕᐅᓯᖅᑐᐃᓐᓇᕐᒥᒃ ᖃᐅᔨᒪᓇᖅᑐᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᖅᑕᖅᑐᒥᒃ M'Clintock ᓯᐸᓗᒧᑦ, ᐅᖅᑲᖅᓯᒪᔪᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᐅᔪᖅ M'Clintock ᓯᐸᓗᒧᑦ ᓯᖁᓯᐊᑎᙳᕆᕆᐊᖅᑐᑦ ᓇᓄᐃᑦ ᑕᐃᑲᓂ. ᑕᒪᑐᒥᙵᑦ ᐊᐃᑉᐱᐊᑎᖅᑲᐃᓐᓇᖅᑐᖑ PBTC-ᑯᓐᓄᑦ ᐊᒻᒪᓗ PBAC-ᑯᓐᓄᑦ, ᓇᓄᐃᑦ ᐱᓕᕆᐊᒍᓂᖏᓐᓄᑦ ᑲᑎᒪᔨᐊᓛᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᑎᒪᔨᐊᓛᑦ. ᐃᓯᒪᓕᐅᑉᑯᑦ, ᑎᑎᖅᑲᖅᑕᖅᑲᖅᓱᐊᖅᑎᓪᓗᒍ ᓯᖅᓗ, ᑐᓵᕙᐅᔪᖅ ᐃᒪᓐᓇᐃᓐᓂᐊᖅᑐᓂ, ᐱᔭᒃᓴᐅᑎ ᐊᒥᓲᖑᕆᓐᓂᖅᓴᐅᑎᓕᖅᑐᖅ ᓇᓄᐃᑦ ᐅᖃᐅᓯᐅᓂᖏᑦ ᓯᖁᓯᐊᑎᙳᕆᓪᓗᑎᒃ. ᐱᓕᕆᐊᖅ ᒪᓕᒃᑕᐅᔪᓇᐊᓕᒃ, ᒪᓕᒃᑕᐅᓗ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎ, ᒪᓕᒃᑕᐅᖏᑦᑐᖅ. ᐅᖅᑲᐅᓯᓇᔭᖅ ᑕᐃᒪᓐᓇ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᒃ ᐱᔾᔪᑎᒋᓗᒋᑦ ᑲᑎᒪᓂᖅᑕᖅᑲᖅᓂᐊᖅᐸᑦ, ᐃᓚᖑᓂᑦ ᐅᖅᑲᐅᓯᐅᑎᐅᔪᓂᑦ ᐱᔾᔪᑎᖅᑲᓇᐊᖅᑐᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔭᒃᓴᐅᑎᐅᔪᖅ ᓯᓇᐅᖑᒪᖑᕕᑦ.
- 38-ᖑᓚᐅᖅᑎᓪᓗᒋᓪᓗᐊᕐᓂᑦ, ᐊᖏᔪᒥᒃ ᐃᑲᔪᖅᓯᒪᔾᔭᒃ ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑕᐃᑉᓱᒪᓂ. ᑕᐃᑉᓱᒪᓂ, ᐅᖅᑲᓪᓚᖅᑲᐅᑎᓕᓚᐅᖅᑕᒃᑲ ᑲᑎᒪᔩᑦ ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐱᖁᔨᓚᐅᖅᒪᑕ 45-ᓂᒃ (ᓇᓚᐅᑦᑖᖅᑕᐅᓚᐅᖅᑐᖅ 842-ᖑᓚᐅᖅᑐᑦ, ᑕᐃᑲᙵᑦ 4.5%-ᖏᑦ 842-ᖑᔪᓂᑦ 38-ᖑᓚᐅᖅᑐᑦ), 45-ᓂᒃ ᐱᖁᔨᓚᐅᖅᑐᒍᑦ ᐊᒻᒪᓗ ᐅᖅᑲᖅᓱᑕ ᐊᑐᖅᓱᒋᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ, ᐃᓯᒪᔪᒧᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ ᐊᒥᓲᓂᖅᓴᐅᔪᙳᐊᑦᑎᐊᖅᑐᑦ 842-ᖑᔪᓂᐅᖑᓂᑦ ᐊᒻᒪᓗ ᓇᓗᓇᙱᓐᓂᖅᒦᓐᓇᔭᖅᑐᖅ ᓯᓚ ᐊᒥᓲᖏᓕᓪᓚᑦ ᐃᓚᓯᕗᖅᐸᓗᓂ ᐅᕙᖕᓂᑦ ᐋᖅᓯᕗᖅᐸᓗᓂ ᓇᓚᐅᑦᑖᖅᑕᐅᓗᓂ ᖃᐅᔨᓇᓱᒃᑐᓕᕐᓂᐅᑉ. ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᐱᖁᔨᑉᓗᑎᒃ ᐃᓯᒪᓕᐅᓚᐅᖅᑐᑦ 38-ᒥᑦ, 4.5%-ᖑᔪᖅ 842-ᓂᑦ. ᑕᐃᑉᓱᒪᓂᖅ, ᐊᐱᕆᓚᐅᖅᑕᒃᑲ ᑲᑎᒪᔩᑦ, ᐃᑲᔪᓚᐅᖅᑕᒪ ᐱᓕᕆᐊᖑᑎᓪᓗᒍ ᑎᑎᖅᑲᖅᓯᒪᔪᑦ ᑎᑎᖅᑲᐅᑏᑦ, ᐊᐃᑉᐱᐊᒪᖑᒪᖑᕕᑦ, ᐊᒥᓲᔪᖕᓂᕆᐊᖅᑐᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ ᐊᒻᒪᓗ ᑲᑎᒪᔩᑦ ᐅᖅᑲᓚᐅᖅᑐᑦ ᑐᒃᓯᕋᐅᑕᐅᔪᖅ ᓇᓄᐃᑦ ᓯᖁᓯᐊᑎᓇᓂᐊᖏᓪᓗᑎᒃ ᐊᒥᓲᑎᒋᓂᖏᑦ, ᑭᓯᐊᓂ ᑭᖑᓂᐊᒍᑦ ᐅᖅᑲᓪᓚᖅᑲᐅᑎᓇᓂᓕᓚᐅᖅᑐᑦ. ᐃᓯᒪᓕᐅᖅᑕᖅᓴ ᐱᔭᕇᖅᑲᖅᒪᑦ ᓯᖁᓯᐊᑎᖅᑯᔭᐅᖏᖅᓱᒋᑦ. ᐊᒻᒪᓗ ᐊᓯᐊᒍᑦ, ᐃᓄᐃᑦ ᐅᖅᑲᐅᓯᖅᓱᒋᑦ

ᐊᒥᓲᔪᖅᓃᕆᐊᕐᓗᒋᑦ ᐅᕝᕙᓘᓐᓂᑦ ᑐᕌᒐᖃᕐᓗᑎᒃ ᐊᒥᓲᔪᖅᓃᕆᐊᕐᓂᐊᕐᓗᒋᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖅᑎᑦ. ᑕᐃᒪᓐᓇ ᑕᒡᕙ ᑐᓵᖃᑦᑕᓚᐅᖅᑐᖓ ᐊᑐᖅᓯᒪᔭᒻᓂᑦ. ᐅᖃᕈᖕᓇᖅᑐᖓ ᑲᑎᒪᔩᑎᒋᔭᐅᓕᖅᐸᑦ, ᐅᖃᓪᓗᕆᒃᓴᐅᑎᒋᓗᒋᑦ ᓇᓗᓇᙱᑦᑐᖅ ᐱᔭᒃᓴᐅᑎᐅᔪᖅ ᓇᓄᕐᓄᑦ ᐱᒻᒪᕆᐅᓇᔭᖅᑐᖅ ᐃᓚᐅᑎᓐᓂᐊᕐᓗᒋᑦ ᐅᖃᓪᓗᕆᒃᓴᐅᑎᓂᕐᓂᑦ.

- **ᔪᐋᓐ** ᐅᖃᖅᑐᖅ ᖃᓄᖅ ᐊᒥᓱᐊᓂᒃ ᓇᓐᓄᒡᓗᓂ ᐊᒻᒪᓗ NDF-ᑖᕐᓗᓂ ᐱᑦᑕᐅᖅᑎᑦᑑᓇᔾᙱᒪᖖᑦ ᓇᓄᐃᑦ ᐊᒥᖅᑎᑦ ᐱᔭᐅᔪᒪᓗᐊᕈᖅᓃᖅᑎᓪᓗᒋᑦ.
- **ᐊᕙᓐ**: 100%-ᒥᒃ ᐊᖏᕐᑲᑕᐅᔪᖅᓚ ᐊᑦᑕᕐᓇᖅᑐᒦᑎᑦᑎᙱᓐᓂᕐᒧᑦ ᐱᕝᔪᑕᐅᔪᓗᒃᑖᓂᒃ ᐊᒻᒪᓗ ᐃᓄᖕᓄᑦ ᐊᑐᖅᑕᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᒃᐱᒋគᐅᓂᕐᑲᕐᓂᖅᓚᓄᑦ ᑕᒪᓐᓇ. ᐱᒻᒪᕆᐅᔪᖅ ᐅᖃᓪᓗᕆᒃᓴᐅᑎᕐᑲᕐᓗᓂ ᐱᕝᔪᑎᕐᑲᖅᑐᒥᒃ, ᐱᖅᑯᓯᕆᔭᐅᔪᖅ ᐱᑕᕐᑲᕐᓗᓂ ᓇᓄᕐᓄᑦ ᓄᓇᒥᑦ, ᓱᓇᐅᕙ ᑐᕌᒐᐅᔪᖅ ᑕᕝᕗᖅᓚ. CITES-ᒥᑦ, ᓇᓄᐃᑦ ᐊᒥᖅᑎᓐᓄᑦ ᐊᑭᐅᔪᑦ ᐊᑭᓗᒃᑐᑯᓘᓕᖅᑐᑦ ᒫᓐᓇᐅᔪᖅ. ᑦᓴᐃᓇ ᓂᐅᕕᖅᑐᒻᒪᕆᐅᕐᑲᑦᑕᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᓂᐅᕕᓗᐊᕈᖅᓃᖅᑐᖅ. ᐃᓕᕽᓯ ᖃᐅᔨᒪᓗᒃᑖᖅᑐᓯ ᒥᖁᐃᑦ ᐊᑭᖅᑎᑦ ᐊᑭᑦᑐᕆᐊᖅᐸᒃᐸᓗᑎᒃ ᐊᒻᒪᓗ ᐊᑭᓗᒃᓯᒋᐊᖅᐸᒃᐸᓗᑎᒃ, ᓄᖅᑲᖅᓚᓕᖅᐸᒃᐸᓗᑎᒃ ᐊᒻᒪᓗ ᐅᑎᕐᑲᑦᑕᖅᑐᑦ. ᐊᑕᐅᓯᐅᔪᖅ ᓇᓄᐃᑦ CITES-ᒥᑦ ᑎᑎᖅᑲᒧᑦ ᐃᓚᐅᓕᑎᔭᐅᔪᖅ 1-ᒧᐊᖅᑕᐅᒃᐸᑦ (ᖁᒻᒧᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ), ᐅᑎᕈᒫᕐᓂᖅᑎᑦ ᐊᑦᑎᒃᑑᔪᖅ, ᑕᐃᑯᖅᓚ ᑎᑭᐅᑎᒍᕙᑦ, ᐊᒥᖅᑎᓐᓄᑦ ᐊᑭᐅᔪᑦ ᐅᑎᕋᓗᐊᖅᐸᑕᓘᓐᓃᑦ, ᑕᐃᑯᙱᓚᕈᕙᑦ CITES ᑎᑎᖅᑲᒧᑦ ᐃᓚᓯᐅᑎᓯᒪᔪᖅ 1-ᒧᑦ, ᐆᒪᔪᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᖅᓃᑖᕐᓂᖅᑎᑦ ᑭᖑᓂᐊᒍᑦ ᑕᒪᑐᒪ ᐊᑦᑎᒃᑯᒻᒪᕆᐅᕐᑲᑦᑕᖅᓯᒪᔪᖅ. ᐅᖃᙱᑦᑐᑦ ᑕᐃᒪᓐᓇᐃᑦᑐᖅᓇᙱᑦᑐᖅ, ᑭᓯᐊᓂ ᐅᔾᔮᖅᑐᕐᓂᕐᒨᑐᐃᓐᓇᖅ. ᑐᑭᓯᐊᔭᒃᑲ ᐅᖃᐅᓯᕆᔭᓯ ᐊᒻᒪᓗ ᒫᓐᓇᐅᔪᖅ ᐊᑭᖅᑎᑦ ᓇᓄᐃᑦ ᐊᒥᖅᑎᑦ, ᐊᒻᒪᓗ ᓄᓇᕐᔪᐊᕐᒥᑦ ᐱᔪᑦ ᓇᓗᓇᙱᑦᑐᑦ ᐊᐃᑦᑖᖑᖅᒪᑦ ᐃᓕᕽᓯᖅᓄᑦ.
- **ᔪᐋᓐ** ᐅᖃᖅᑐᖅ ᓂᐅᕕᐊᒃᓴᖃᕐᓂᖅ ᓇᓄᐃᑦ ᐊᒥᖅᑎᓐᓂᒃ ᐱᖅᑯᓯᕆᔭᐅᔪᒧᑦ ᐱᕕᖃᖅᑎᑦᑎᔪᖅᓇᙱᒋᒪᑕ ᒪᒃᑯᒃᑐᓄᑦ ᐊᒻᒪᓗ ᓂᐅᕕᖅᑕᐅᓗᑎᒃ ᓄᓇᕗᒻᒥᑦ.

- **Hᐃᐅᕆ**: ᐊᒥᓲᙱᕆᐊᖅᑐᓂᑦ ᐃᖏᖅᓂᒃ ᒥᐊᓂᕆᐸᓗᑎᒃ ᑐᖁᑕᐅᔪᓂᒃ?
- **ᔪᐋᓐ**: ᐊᔾᔨᒌᙱᑦᓂᕐᑲᑦᑎᐊᖅᑐᑦ ᓇᓄᐃᑦ ᑲᒪᒋᔭᐅᓂᖅᑎᓐᓂᑦ ᑕᒡᕙᓂ ᑕᐅᑐᖅᑳᖅᓗᒋᑦ ᐊᓯᖅᑎᓐᓂᑦ ᓄᓇᕗᒻᒥᑦ ᐳᓚᕐᑲᖅᑐᓕᕆᓂᖅ ᒫᓂᑖᐸᒥᑦ ᐱᔪᑎᒋᐸᓗᒍ. ᐃᑲᔪᖅᑐᐃᔪᑦ ᐃᓄᖕᓄᑦ ᓄᓇᒃᑰᕈᑎᒦᑦᑐᓂᑦ ᖃᓂᖁᔭᐅᐸᓗᑎᒃ. ᖃᐅᔾᔭᓂᐊᖅᑐᑎᑦ ᓇᓄᐃᑦ ᐃᓄᐃᑦ ᐃᒡᓕᖅᑎᑦᑕ ᐃᓚᓖᖅᑎᑦᑎᒍᑦ ᑕᐅᑐᒃᑐᓂᑦ ᐊᕐᕕᐊᓂᑦ. ᑕᐃᒪᓐᓇᐃᑦᑐᓂᑦ ᖃᐅᔮᔮᖅᑎᑦᑐᒍᑦ ᐃᒃᐱᐊᕐᔪᖅᒥᑦ, ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᑲᐸᐊᓱᒃᑐᑦ ᐃᓄᖅᓂᑦ. ᐳᓚᕋᖅᑐᓕᕆᓂᖅ ᐱᕝᔪᑕᐅᔪᖅ ᓇᓄᐃᑦ ᑲᐸᐊᓇᓗᐊᕐᓂᖅᑎᓐᓄᑦ.

- **Hᐃᐅᕆ**: ᓈᒻᒪᒋᓂᖅ ᐊᔪᖅᑕᕋ ᖃᐸᓯᐅᓂᖅᑎᓐᓂᑦ, ᐅᖃᐅᓯᐅᔪᑦ ᐊᑐᖅᑕᐅᐸᓗᓂ ᓈᓴᐃᓂᕐᒧᑦ ᐋᖅᑭᐅᒪᔪᖅ. ᐊᔪᖅᑕᕋ ᓈᒻᒪᒋᓂᐊᕐᓗᒋᑦ ᐅᕝᕙᑎᖅᓄᑦ ᑐᓂᐅᔭᔪᑦ ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖅᓇᖅᑐᑦ, ᑭᓯᐊᓂ ᖃᓄᐃᓕᐅᕐᕕᖅᑲᙱᑦᑐᒍᑦ. ᑐᓂᔭᐅᖅᑲᑦᑕᖅᑐᑦ ᓄᓇᕗᒥ ᒐᕙᒪᒃᑯᓐᓂᑦ ᑐᓂᔭᐅᔪᓂᒃ. 20-30 ᐅᑭᐅᓃᓕᖅᑐᖅ, ᐃᓐᓇᕆᔭᕗᑦ ᐅᖃᓪᓚᖅᑲᑦᑕᓚᐅᖅᑐᑦ ᑕᒪᑐᒥᙱᓚᑦ, ᐊᑐᖅᑕᐅᔪᖅ ᓇᓚᐅᑦᑎᙱᑦᑐᖅ. ᓈᓴᐅᑎᓂᒃ ᑐᓂᔭᐅᖃᑦᑕᖅᑐᒍᑦ ᓇᓚᐅᑦᑎᙱᑦᑐᓂᒃ. ᑕᐃᒪᓐᓇᒧᑦ ᑕᐃᒪᓐᓇᓯᓕ ᐅᖃᖃᑦᑕᐃᓐᓇᖅᑐᒍᑦ.

- **ᑭᓖᑕᓐ**: ᐱᒋᐊᕈᖅᓇᖅᐱᑕ ᖃᓄᐃᖅᓚᓂᐅᔪᓂᑦ ᑎᖏᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᓂᒃ ᐊᒻᒪᓗ ᐅᖃᐅᓯᕆᓗᒋᑦ ᑕᑯᔭᐅᔪᖅᓇᖅᑐᑦ? ᒪᕿᒃ ᐅᐸᓘᖅᓄᑦ ᓈᓴᐃᑎᓪᓗᒋᑦ, ᖃᐅᔨᒪᔪᖅᓚ ᐱᑕᖃᖅᑐᖅ ᑕᑯᙱᑕᐸᑎᖅᓂᑦ. ᐊᒻᒪᓗ, ᑎᖏᓲᒃᑯᑦ ᐃᒥᓱᐃᖅᑕᖅᓯᒪᔪᖅᓚ ᐊᒻᒪᓗ ᐊᑐᖅᑕᐅᔪᑦ ᖁᑦᑎᖕᓂᐅᔪᖅ ᓈᓴᐃᓂᐊᕐᓗᓂ ᐊᑦᑎᓗᐊᖃᑦᑕᖅᑐᑦ ᑕᑯᔪᖅᓇᑦᑎᐊᕐᓂᐊᕐᓗᒋᑦ. ᖁᑦᑎᖅᓂᖅᓴᐅᒍᐸᓯ, ᑕᑯᔪᖅᓇᕐᓂᖅᓴᐅᓇᔭᖅᑐᒃᓴᐅᔪᓯ ᓇᓄᕐᓂᑦ. ᑎᖏᓲᕆᔭᑉᑯᑦ ᒫᓂᑑᐸᒧᙱᓚᐅᖅᑲᑦᑕᖅᓯᒪᔪᖅᓚ ᐊᒻᒪᓗ ᑕᑯᓯᒪᔪᖅᓚ ᓇᓄᕐᓂᑦ ᐸᖅᓚᓪᓚᒃᑐᑦ ᐱᕈᖅᑐᓄᑦ ᑕᓕᒋᐊᖅᐸᓗᑎᒃ ᑕᑯᔪᖅᓇᕐᓂᐊᕖᖅᑎᓪᓗᒋᑦ. ᑕᐃᒪᓐᓇᐃᖅᑲᑦᑕᖅᑐᑦ ᓈᓴᐃᑎᓪᓗᑎᑦ, ᑕᑯᙱᑦᑐᖅᓇᖅᑐᒧᑦ ᓈᓴᐃᑎᓪᓗᑕ ᓇᓄᕐᓂᑦ.
- **ᐊᓕᓴ**: ᖁᔭᓐᓇᒦᒃ ᑕᒪᓐᓇ ᐅᖃᐅᓯᕆᓚᖅᓂ. ᑐᑭᓯᐊᔪᖅᓇᖅᑐᖅᓚ ᐱᕝᔪᑕᐅᔪᑦ ᑕᑯᔭᐅᔪᖅᓇᕐᓂᖅᑎᓐᓄᑦ 4.5%-ᒥᒃ ᐊᑐᐃᓐᓇᖅᑲᑦᑕᖅᑎᓪᓗᑕ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᓂᑦ. ᖃᐅᔨᒪᔪᖅᓚ ᖃᖅᓚᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖅᑎᓐᓂᒃ ᓈᓴᐃᓂᖅ ᐱᖁᔭᐅᓕᓵᕐᒪᑕ, ᐱᔭᐅᓂᐊᕐᓗᑎᒃ ᖃᓄᐃᓕᐅᖅᐸᓪᓕᐊᓂᖅᑎᓗᐊᖅ.

- **ᕈᐃᒪᓐ**: ᐅᖃᖅᖃᐅᓚᕕᑦ ᐱᑦᑕᐅᔫᓂᕋᖅᓗᒍ ᑕᒪᓐᓇ ᐊᒻᒪᓗ ᐅᖃᖅᓗᑎᑦ ᐱᔭᐅᖃᑦᑕᖅᑐᑦ ᐱᔭᐅᔪᒪᔭᖅᑎᑦ ᐆᒪᔪᓕᕆᔨᐊᑏᒃᑯᓐᓄᑦ ᐊᒻᒪᓗ ᓄᓇᓕᖅᒥᐅᓄᑦ. ᒫᑲᔅ ᐊᔾᔨᖅᑎᓐᓂᒃ ᑎᑎᕋᐃᓚᐅᖅᑐᖅ. ᒥᑦᓴ ᐆᓗ ᐊᔾᔨᖅᑎᓐᓂᒃ ᑎᑎᕋᐃᓚᐅᖅᑐᖅ. ᑭᓯᐊᓂ ᓱᓕ ᐊᑐᖅᑕᐅᖅᑎᓐᓇᖅᑐᖅ ᓱᓕ ᐅᐸᓗᒥᒧᑦ ᑎᑭᕐᓗᒍ. ᖃᐅᔨᒪᔪᖅᓚ ᑕᒪᐃᐸᓯᒃ ᓄᑖᖑᓚᐸᓯ ᐊᒻᒪᓗ ᐱᒋᐊᓕᔮᖅᓗᓯ. ᖃᓄᐃᖅᓚᓂᕆᓂᐊᖅᑕᖅᓚ ᑲᑎᒪᓂᐅᑉ ᑕᐃᒪᑐᖅ ᐊᑐᖅᑕᐅᓂᐊᖅᐳᖅ. ᓂᒃ ᓚᓐ ᑐᓴᓚᐅᖅᑎᑦᑐᖅ, ᐅᖃᑕᐅᓯᔭᐅᓚᐅᖅᑐᖅ ᑲᑎᒪᔫᓄᑦ ᓄᓇᓕᖅᒥᐅᓂᑦ ᐅᖃᖃᑎᖃᖅᑯᔭᐅᐸᓗᓂ. ᐱᓚᐅᖅᑎᑦᑐᖅ. ᐊᓕᐊᓇᐃᒋᔭᒋᑦ ᑕᐃᒪᓐᓇᐃᑦᑐᖅᓇᕋᕕᑦ, ᐊᕙᓐ. ᓇᓗᓇᐃᖅᓯᓂᑦᑎᐊᕙᐅᔪᖅ. ᐅᑭᐅᒃᑲᓪᓚᖅᓄᑦ, ᐃᓄᐃᑦ ᐊᒃᑐᖅᓯᖅᑲᑦᑕᖅᑯᔾᔨᓯᒪᙱᑦᑐᑦ, ᓄᑮᕈᑎᓐᓇᖅᑐᓂᒃ ᐊᑐᖅᑕᐅᖅᑯᔾᔨᙱᕋᓗᑎᒃ, ᑖᒡᓕᖅᑐᐃᖅᑯᔾᔨᙱᕋᓗᑎᒃ, ᖁᖓᓯᕈᓯᖅᑐᐃᖅᑯᔾᔨᙱᕋᓗᑎᒃ. ᓂᕿᖅᑎᓐᓂᒃ ᐲᖅᓯᕝᕕᐅᔪᑦ ᐊᖏᔪᒥᒃ ᐱᒋᐊᕐᓂᐅᔪᖅ ᓯᕗᒧᒋᐊᕐᓂᐊᕐᓗᑎᒃ ᐃᓄᐃᑦ ᑐᓵᓚᐅᓯᐅᒃ. ᑕᒪᓐᓇ ᐊᓕᐊᓇᐃᒋᔭᕗᑦ. ᐱᔭᐅᔪᒪᙱᑦᑐᑦ ᓇᓄᕋᐃᑦ.

- ᑕᐃᔭᐅᕙᓚᐅᖅᑐᑦ ᓇᓄᕐᓄᑦ ᑎᑭᐅᑎᔪᑦ. ᒫᓇ ᖃᑉᓯᐅᓂᖏᑦ ᓇᓄᐃᑦ ᐃᓘᐃᓗᑕᐅᔪᓂᑦ ᐱᑕᖃᕐᓂᐊᖅᐱᑕ? ᐊᑦᑕᕐᓇᖅᑐᒦᑎᓐᓂᐊᖏᓪᓗᒋᑦ ᓄᓇᕗᒻᒥᐅᓗᒃᑖᑦ. ᓄᓇᑦᑎᐊᖑᓚᐅᖅᑎᓪᓗᒍ, ᓇᓄᐃᑦ ᓈᓴᐅᓯᖅᑐᖅᑕᐅᕙᓚᐅᖅᑐᑦ ᑰᖕᔪᐊᕐᒥᑦ, ᒫᓇᓕ ᑕᑯᔪᖕᓇᙱᓕᖅᑕᕗᑦ, ᑕᐃᒪᓐᓇ ᐃᑲᔪᖅᑐᖅ. ᓯᕗᒻᒧᒃᑉᓪᓕᐊᔪᖅ ᓱᑲᐃᑦᑐᒥᑦ. ᑐᒃᓯᕋᐅᑕᐅᔪᑦ ᐊᒥᓲᔪᖕᓃᖅᑕᐅᕐᑯᑉᓗᒋᑦ ᑐᓵᖅᑎᑦᑎᔾᔪᑕᐅᕐᑲᑦᑕᖅᓯᒪᔪᖅ ᐊᑕᐅᓲᙱᕐᓗᒍ ᑕᐃᒪᐅᖅᖄᖅᑎᓐᓇᓯ, ᒫᑲᔅ ᑕᐃᒪᐅᖅᖄᖅᑎᓐᓇᒍ ᐊᒻᒪᓗ HACCS ᐋᖅᑭᐅᒪᔪᖅ, ᐱᑕᖃᕆᖅᓃᕆᐊᖃᖅᑐᖅ. ᓇᓂᑐᐃᓐᓇᖅ ᐊᔪᖅᑐᖅ. ᓇᓂᑐᐃᓐᓇᖅ ᓄᓇᕐᔪᐊᕐᒥᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᙱᑦᑐᖅ. ᓲᖅ ᐊᑐᖅᑕᐅᓕ ᓄᓇᕗᒻᒥᑦ, ᓇᓄᕐᓄᑦ?
- ᑐᓴᕐᓂᐊᖅᑐᓯ ᐊᒥᓲᓂᒃ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᖏᓐᓂᑦ ᐃᓄᐃᑦ ᐊᒻᒪᓗ ᐱᕐᑯᑎᒋᔭᐅᔪᑦ. ᑐᓵᓂᐊᖅᑐᓯ ᐊᒥᓲᓂᒃ, ᓇᒧᑐᐃᓐᓇᖅ ᑎᑭᒃᑲᓗᐊᕈᑉᓯ.

- **ᑭᓖᑕᓐ**: CITES ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓄᑦ, ᑭᑐᓐᓄᑦ ᖃᐅᔨᔭᐅᕐᑲᑦᑕᖅᐸᑦ ᐋᖅᑭᒃᑕᐅᓂᖏᑦ ᐊᑐᖅᑕᐅᔪᑦ CITES-ᓄᑦ ᐱᓕᕆᐊᖑᔪᑦ? ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔩᒃᑯᓐᓄᑦ ᐱᓕᕆᐊᖑᕐᑲᑦᑕᖅᐸᑦ?
- **ᐊᕙᓐ**: ᐋᖅᑭᒃᓱᖅᑕᐅᓯᒪᔪᑦ ᐱᓕᕆᐊᖑᕐᑲᑦᑕᖅᑐᑦ ᑲᑎᒪᔩᐊᓛᒃᑯᓐᓄᑦ ᖃᓄᐃᖓᓂᖏᓐᓄᑦ ᐅᓗᕆᐊᓇᖅᑐᒦᑦᑐᓄᑦ ᓂᕐᔪᑎᓄᑦ ᑲᓇᑕᒥᑦ (COSEWIC) ᓇᓄᕐᓄᑦ. ᒫᓇ, ᑲᓇᑕ ᐊᑕᐅᓯᐅᑉᓗᓂ ᖃᓄᐃᑦᑑᖓᔪᓕᐊᕆᔭᐅᔪᖕᓇᖅᑑᑉᓗᓂ. ᑕᒪᕐᒥᓗᒃᑖᖅ 13-ᖑᔪᑦ ᐃᓂᐅᔪᓂᑦ ᐊᑕᐅᓯᐅᑉᓗᓂ ᓇᓄᐃᑦ ᖃᓄᐃᑦᑑᖓᔪᓕᐊᖑᔪᖕᓇᖅᑐᑦ.
- **ᑭᓖᑕᓐ**: ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᒃᑐᖅᓯᓂᖃᕐᓂᐊᖏᑦᑐᑦ CITES-ᒥ?
- **ᐊᕙᓐ**: ᖃᓄᐃᓕᐅᖅᑎᑦᑎᓂᖃᓕᑐᐃᓐᓇᕆᐊᓕᒃ ᐃᓄᖕᓄᑦ ᑕᑯᔭᐅᓗᓂ ᐊᔪᙱᓐᓂᖓᓄᑦ ᐊᔪᕐᓂᐊᕐᓂᖓᓄᓪᓘᓐᓃᑦ. ᐱᖅᑯᓯᕆᔭᐅᔪᖅ ᐊᔪᙱᑦᑐᖕᓇᕐᓂᖓᓄᐊᖃᕐᑲᑦᑕᖅᑐᑦ ᖃᑉᓯᓂᒃ ᓇᓄᕐᓂᑦ ᓄᓇᒥᑦ ᐱᑕᖃᕐᑯᔭᐅᖕᒪᖔᑦ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᓇᓄᐃᓪᓗ ᑎᑭᐅᑎᔾᔪᑎᖃᕐᑲᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ, ᐊᓯᐊᙳᖅᑐᓂᑦ ᑕᕆᐅᑉ ᓯᑯᖓᓂᑦ. ᖃᐅᔨᒪᙱᑦᑐᒍᑦ ᐃᓂᖑᒧᑦ ᖃᑉᓗᓈᑦ ᓄᓇᖓᓃᑦᑐᒥᑦ ᐱᔭᕆᐊᖃᖅᑐᒥᒃ ᐱᓂᐊᕐᒪᖔᑦ ᐊᒻᒪᓗ ᐅᕙᖓᐅᔫᖏᒪᒪᑦ. ᐱᕐᑯᔩᓗᖓᐅᖏᑦᑐᖅ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓄᐊᖅᑕᐅᓗᑎᒃ, ᐃᑲᔪᖅᑐᙱᑦᑕᕋ ᑕᒪᓐᓇ. ᐃᓯᒪᔪᖓ ᒪᕐᑲᐃᑕᐅᔪᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐋᖅᑭᐅᒪᔪᖅ ᐊᔪᙱᒪᓪᓚᑦ ᐊᒻᒪᓗ ᐅᒃᐱᕈᓱᒃᑐᖓ ᐱᖅᑯᓯᕆᔭᐅᔪᒃᑯᑦ ᐊᑐᖅᑕᐅᓂᖏᑦ ᐱᕝᕕᐅᕐᑲᑦᑕᖅᑐᑦ. ᑐᑭᓯᐊᑦᑎᐊᖅᑐᖓ ᑕᐅᑐᒐᕆᔭᐅᔪᒥᒃ ᓇᓄᖅᑕᕐᑲᒃᑲᓐᓂᕐᑯᓴᙱᑦᑐᓂᒃ ᓄᓇᒥᑦ. ᐱᓕᕆᓯᒪᔪᖓ ᓇᓄᕐᓂᑦ. ᑎᑭᐅᑎᓯᒪᕐᑲᑦᑕᕐᓂᑰᔪᖓ ᐊᒻᒪᓗ ᖃᐅᔨᒪᔪᖓ ᖃᓄᖅ ᑲᐸᐃᐊᓇᖅᑎᒋᓂᖓᓂᑦ. ᐃᓄᐃᑦ ᐃᓅᔪᖕᓃᖅᓯᒪᔪᑦ ᐊᒻᒪᓗ ᑕᒪᓐᓇ ᐊᐃᑦᑖᒻᒪᓇᐊᓘᔪᖅ. CITES-ᑕᓇᑎᓪᓗᑕ, ᐃᓄᒃᑕᕐᑲᖅᑐᖅ ᓂᒋᐊᓂᑦ ᐋᕗᓇᑲᒥᑦ ᐅᕐᑲᖅᑐᓂᑦ ᑎᑎᕋᖅᑕᐅᑎᑦᑕᓇᐊᕐᑲᕋᑉᑎᒍᑦ ᓇᓄᐃᑦ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓ ᐱᔾᔪᑎᒋᐸᓗᒍ, ᑕᐃᒪᓐᓇᒎᑦ ᖃᐅᔨᒪᓐᓇᙱᑦᑐᖅ ᑭᑑᓂᐊᕐᒪᖔᑕ ᐱᓕᕆᔪᓂᒃ ᐊᒻᒪᓗ ᑐᒃᓯᕋᐅᑕᐅᔪᑦ ᓴᖅᑭᑎᑕᐅᓗᑎᒃ, ᑕᐃᒪᓐᓇᒎᑦ ᑕᒪᓐᓇ ᐊᔪᕐᓇᖅᑑᔪᖅ.

- **Hᐃᐅᓇ**: ᐊᐅᔭᐅᓚᐅᖅᑐᒥᑦ, ᐆᒪᔪᑕᓇᕐᒃᑯᑦ ᖃᐅᔨᒋᐊᓚᐅᖅᑐᑦ ᕿᓇᐅᔭᖅᑲᖅᑎᑕᐅᔪᖕᓇᕐᒪᖔᑕ WWF-ᑯᓐᓂᑦ ᓇᓄᕐᓂᑦ ᑎᑭᑦᑕᖅᑐᓄᑦ ᒥᐊᓂᖅᓯᔨᐅᓂᐊᖅᑐᓄᑦ, ᑭᓯᐊᓂ ᓯᕗᓪᓕᖅᐹᑦᑎᐊᖑᓚᐅᕐᒪᑦ ᐅᖃᐅᓯᕆᔭᐅᓂᖓ ᑕᒪᓐᓇ ᐊᒻᒪᓗ ᐱᑕᖃᕆᐊᖃᓪᓚᑦᑖᕐᓂᖓᓂᒃ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᕐᓂᐊᖅᑐᓂᒃ. ᒪᕿᒃ ᐃᓄᐃᑦ ᐱᓕᕆᑖᖑᓂᐊᖅᑐᓄᑦ ᐊᐅᔭᑕᒫᒃᑯᑦ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑎᑐᐃᓐᓇᐅᓂᐊᕐᓗᑎᒃ ᐅᕝᕙᓘᓐᓃᑦ ᓄᓇᓕᖕᒥᑦ, ᑰᖕᓂᑦ, ᐊᒻᒪᓗ ᑕᖕᒫᕐᕕᐅᔪᓂᑦ ᐊᒻᒪᓗ ᐃᕐᑲᓪᓚᐊᕐᕕᐅᓯᒃᑐᓂᑦ, ᐊᒻᒪᓗ ᐃᒪᓐᓇᐃᑦᑐᒍᑦ ᓱᓕ, ᐃᓱᒪᔪᖓᓕ, ᖃᐅᔨᓴᐃᓂᒃ ᐱᑕᖃᕆᐊᖃᖅᑐᖅ. ᐱᔭᐅᔭᓇᐊᕐᑲᑦᑎᐊᖅᑐᖅ ᒫᓇᐅᔪᖅ ᐊᒥᓱᒻᒎᖅᓯᒪᖕᒪᑕ ᓇᓄᐃᑦ ᖃᓂᒋᔭᑉᑎᖓᓂᑦ ᐅᐱᕝᕚᑕᒫᒃᑯᑦ, ᐊᐅᔾᑯᑦ, ᐊᐅᔭᓗᒃᑖᖅ ᐅᑭᐊᒃᓵᒍᑦ ᑎᑭᕐᓗᒍ.

- **ᔪᐋᓐ** ᐅᖃᖅᑐᖅ ᐃᓱᒪᒋᔭᐅᔪᒥᒃ ᓱᓕᖕᒋᑦᑐᒥᒃ ᓇᓄᐃᑦ ᐱᑦᑕᑦᑕᕐᑲᑦᑕᖅᑐᑦ ᐊᐅᔮᒃᑯᑦ – ᕿᓇᓗᖓᖅᑕᐅᓚᐅᖅᑐᓂᑦ ᓂᓇᔭᒃᓴᖃᕐᑲᑦᑕᖅᑐᑦ. ᐃᖏᖕᓂᒃ ᒥᐊᓂᓇᑉᓗᑎᒃ ᑐᕐᑯᑕᐅᔪᑦ ᓇᓄᐃᑦ ᖁᐃᓂᑦᑎᐊᖃᕐᑲᑦᑕᖅᑐᑦ.

- **Hᐃᐅᓇ**: ᖃᐅᑕᒫᑦ ᑕᑯᖃᑦᑕᖅᑐᖓ ᓴᖅᑭᑕᐅᔪᓂᒃ ᓱᒋᐊᖅᑎᑦᑎᓇᓱᒃᑎᓂᒃ ᐊᒻᒪᓗ ᑕᒪᓐᓇ ᐅᑉᓗᒥᐅᔪᖅ, ᑭᓯᐊᓂ ᖃᐅᑕᒫᑦ ᓴᖅᑭᑦᑎᖃᑦᑕᖅᑐᑦ ᓇᓄᐃᑦ ᒥᒃᓵᓄᑦ, ᓱᓕᖖᒋᑦᑎᐊᖅᑐᖅ.

- **ᑖᓇ**: ᐅᖃᒃᑲᓐᓂᕐᓗᖓ ᐊᔾᔨᒌᖖᒋᑦᑐᑦ ᒪᓕᖓᓂᒃ ᐃᓐᓇᓇᔭᑉᑕ ᒪᓖᓐᓇᖃᕐᑲᑦᑕᖅᑕᖏᑦ. ᐃᓗᐃᓅᖅᑕᐅᖏᓐᓇᖃᕐᑲᑕᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ ᐊᓕᐊᓇᐃᖏᑦᑐᑐᐃᓐᓇᐅᖏᑦᑐᖅ, ᑭᓯᐊᓂ ᑕᖃᔪᑦ ᐃᓄᐃᑦ ᑕᒪᐅᖓ ᑎᑭᖃᑦᑕᖅᑐᓂᒃ, ᓈᓴᐃᔪᔪᓂᒃ ᐅᕝᕙᓘᓐᓃᑦ ᑕᑯᔭᖅᑐᐃᓚᐅᐱᓪᓚᑐᐃᓐᓇᖅᓱᑎᒃ, ᐅᑭᐅᖅ ᐃᓚᖓᓂᑦ ᐊᒻᒪᓗ ᑕᐅᓄᖓ ᐅᑎᖅᓱᑎᒃ. ᑭᑐᓗᒃᑖᑦ ᐆᒃᑐᕐᑯᒍᑉᑎᒍᑦ ᓴᙱᓂᖃᖅᑐᑦ ᐃᓯᒪᓕᐅᓇᓂᕐᒧᑦ, ᓲᕐᓗ, ᓇᓅᑎᑉᑕ ᒥᒃᓵᓄᑦ, ᐆᒃᑐᕐᑯᒍᑎᒍᓪᓕ ᖢᒫᓇᐊᖅᑐᕐᓗᑎᒃ ᓄᓇᒧᑦ ᐊᒻᒪᓗ ᑕᑯᓂᐊᖅᑐᑎᑦ. ᑕᒫᓂᕐᒥᐅᑕᐅᖏᒃᑯᕕᑦ, ᐱᓯᒪᙱᑕᐃᑦ ᐱᒻᒪᕆᐅᔪᖅ. ᑕᒫᓂᕐᒥᐅᑕᐅᔭᕆᐊᖅᑲᖅᑐᑎᑦ, ᖃᐅᔨᒪᓕᕐᓂᐊᕐᓗᒍ. ᑕᒫᓂᕐᒥᐅᑕᐅᓯᒪᓕᕋᓗᐊᕈᕕᓪᓘᓐᓂᑦ ᐊᕙᑎᑦ ᐅᑭᐅᓄᑦ, ᖃᐅᔨᒪᓕᕐᓗᒃᑖᒃᔫᖏᑕᐃᑦ ᓱᓕ. ᐊᔪᖅᑐᒍᑦ. ᖃᐅᔨᒪᓕᕐᓗᒃᑖᕐᓂᖅ ᐊᔪᖅᑕᕗᑦ. ᐃᓐᓇᓇᔭᐅᔪᑦ ᐅᖃᐅᔾᔫᖏᑦᑐᑦ ᖃᐅᔨᒪᓗᒃᑖᕋᑉᑎᒍᑦ, ᑭᐅᔪᑎᖃᕋᑉᑕ. ᐅᖃᓪᓚᖃᑦᑕᖅᑐᑦ ᐊᑐᖅᓯᒪᓂᑯᖏᓐᓂᑐᐃᓐᓇᖅ. ᐅᖃᓪᓚᖏᒃᑲᓗᐊᖅᐸᑕᐅᓐᓂᑦ ᖃᐅᔨᒪᓂᕐᖅᑑᖓᑕ. ᑭᓇᑐᐃᓐᓇᖅ ᐆᒃᑐᔾᔪᖏᑦᑐᕐᑐᓐᓂᑦ, ᐱᓗᐊᖅᑐᒥᑦ ᑕᒫᓂᕐᒥᐅᑕᐅᖏᒃᑯᕕᑦ. ᑕᒪᓐᓇ ᐊᖏᕐᕌᓗᓂ ᐱᔪᑕᐅᓚᐅᖅᑐᖅ ᐱᓯᒪᐃᓐᓇᖅᓗᓂᔪᖅ. ᐱᐅᒋᖃᑦᑕᓚᐅᖏᑕᖏᑦ ᑕᒪᐅᙳᐅᔪᑦ ᐃᓯᒪᓕᐅᓇᑉᓗᑎᒃ ᐊᒃᑐᖅᓯᑉᓗᓂ ᐃᓅᓯᑉᑎᖏᓐᓂᑦ ᐊᒻᒪᓗ ᐅᑎᓕᖅᓱᑎᒃ ᖃᑉᓗᓈᓄᑦ.

- **ᐊᓕᓴ:** ᖁᔭᓐᒦᒃ, ᑖᓇ, ᑕᒪᓐᓇ ᐅᖃᐅᓯᕆᒐᕐᓂ. ᐃᓱᒪᔪᖓ ᑕᒪᓐᓇ ᑐᖅᒐᓇᖕᒪᔾᔪᒃ ᐊᐅᓚᑦᑎᓂᖅᑕᐅᓂᖏᑦ ᐋᖅᑭᐅᒪᓂᐅᑉ, ᓄᓇᓖᑦ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᐊᒃᑐᖅᓯᓂᖃᕐᕕᒋᔭᕐᒥᑦ, ᑕᐅᑐᒐᕆᔭᓯ ᐃᓚᓕᐅᑎᔭᐅᓂᐊᕐᓗᑎᒃ ᐃᓱᒪᓕᐅᕆᓂᖏᑦ. ᓂᕆᐅᕐᓂᖅᑲᖅᓗᐊᖅᑐᖓ ᐅᖃᓪᓗᓇᒃᓴᐅᑎᓂᖅ ᐱᔭᐅᓯᒪᔪᖅ ᑲᑎᒪᔾᔪᑎᓂᑦ ᐊᒻᒪᓗ ᑐᓂᔭᐅᓂᐊᖅᓱᑎᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ, ᓱᓇᓗᒃᑖᑦ ᑑᓯᔭᐅᔪᑦ ᐅᕙᑎᓐᓂᑦ ᐅᑉᓗᒥ, ᐱᔭᐅᓯᒪᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᒥᑦ. ᓂᕆᐅᒃᑐᖓ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐊᒃᓯᕈᕐᓗᑎᒃ ᐱᓕᕆᖁᑉᓗᒋᑦ ᐃᓚᓕᐅᑎᔭᐅᓂᐊᕐᓗᑎᒃ ᓄᓇᓕᖕᒥᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᑐᓴᖅᑕᐅᔪᑦ. ᐊᔪᕐᓇᖅᑐᖅ ᑭᓯᐊᓂ, ᖃᐅᔨᒪᔪᖓ. ᑭᓯᐊᓂ ᖃᐅᔨᒪᔪᑎᑦ, ᓂᕆᐅᒃᑐᖓ, ᖃᐸᓯᐊᕐᔪᕐᓂᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᓚᐅᖅᑐᓂᑦ, ᐅᕙᓄᑦ, ᓲᕐᓗ ᐃᓚᓕᐅᑎᓂᐊᖅᑕᖏᑦ ᑕᐅᑐᒐᕆᔭᐅᓗᐊᖅᑐᑦ.
- **ᑖᓇ:** ᓱᓇᒥᒃᑭᐊᖅ ᐊᔪᕆᖅᑐᖅᑲᓇᐊᖅᑲᖅᑐᖅ, ᓱᖁᒃᓯᒪᓂᖅᑕᖅᑲᓗᐊᕆᖅᑐᖅ ᑕᒡᕙᓂ ᐃᓅᓯᑉᑎᖕᓂ.
- **ᓅᓪ:** ᖃᑕᙳᑎᑎᑦ ᐅᑐᖁᓇᔭᖅᐱᒋᑦ ᐊᒻᒪᓗ ᓇᓄᖅᓄᑦ ᑎᑭᑕᐅᖅᑲᑕᐅᓗᑎᒃ ᐊᑐᖅᓯᒪᔭᖅᓄᑦ? ᓄᑕᖅᖃᑎᑦ, ᐃᖕᒍᑕᑎᑦ, ᐊᓂᑎᑦ, ᓇᔭᑎᑦ ᐊᒻᒪᓗ ᐃᓐᓇᓇᔭᓯ? ᑭᓇᑐᐃᓐᓇᖅᒧᑦ ᐱᔭᐅᔪᒪᙱᑦᑐᖅ. ᑕᐃᒪᓐᓇᐃᑦᑐᒃᑰᖅᑯᓗᑎᑦ ᐃᓱᒪᓇᔭᙱᑦᑐᒍᑦ, ᐊᒻᒪᓗ ᑭᓇᑐᐃᓐᓇᖅ ᑕᐃᒪᓐᓇᐃᑦᑐᒃᑰᖅᑯᙱᑦᑕᕗᑦ.
- ᖃᐅᔨᒪᔭᓯ ᐊᑐᖅᑲᑦᑕᖅᑕᓯ ᐊᓯᐊᙳᖅᓯᒪᔪᑦ ᐅᑭᐅᑦ ᐊᓂᒍᖅᑐᓂᑦ. ᐱᓕᕆᐊᕐᒍᖅᑲᑦᑕᓕᖅᑐᑦ ᐊᔾᔨᒌᙱᑦᑐᓂᑦ ᐅᑭᐅᑉ ᐃᓗᐊᓂᑦ. ᐱᓕᕆᐊᓕᒍᑉᓯᐅᒃ ᐊᑐᖅᓱᑎᒃ ᐊᔾᔨᒌᙱᑦᑐᓂᒃ ᐊᑐᖅᑕᐅᔪᓂᑦ, ᖃᓄᖅ ᓇᓚᐅᑦᑎᓯᒪᑎᒋᕙᑦ? ᖃᐅᔨᒪᔭᑎᑦ 2011-ᒥᑦ, 2021-ᒥᑦ ᓈᓴᐅᑎᓯ, ᐊᖏᔪᒻᒪᕆᖕᒥᑦ ᐊᔾᔨᒌᙱᓐᓂᖅᑲᖅᑐᑦ, ᑕᐃᒪᓕ ᖃᓄᐃᓕᓂᐊᓕᖅᐸᑦ?

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᖃᓄᐃᓪᓕᓂᐅᔪᑦ

- **ᓴᕗᒃ:** ᓈᓴᐃᖅᑲᑦᑕᖅᑐᑦ ᐊᑯᓪᓕᕈᖅᕕᖕᒥᑦ ᐊᒻᒪᓗ ᐊᒥᖃᐃᔭᖅᕕᖕᒥᑦ. ᐊᒥᓱᐃᑦ ᓇᓄᐃᑦ ᑕᑯᔭᐅᖅᑲᑦᑕᙱᑦᑐᑦ ᐱᕈᖅᑐᐃᑦ ᐃᑉᔪᓂᖅᒋᓐᓄᑦ.
- **ᐊᓕᓴ:** ᑕᒪᒃᑯᐊ ᐃᓱᒪᒋᓇᓱᒃᑲᑦᑕᖅᑕᕗᑦ ᐋᖅᑭᐅᒪᔪᓂᑦ. ᓇᓄᖅᓂᒃ ᑕᑯᔭᕌᖓᑕ, ᑎᑎᖃᐅᓯᐅᖅᑲᑦᑕᕐᒥᔪᑦ ᐊᔾᔨᒌᙱᑦᑐᑦ ᑕᑯᒃᓴᐅᑎᑦᑎᙱᑦᑐᖓᓇᖅᑐᑦ ᐊᒻᒪᓗ ᑕᑯᒃᓴᐅᑎᑦᑎᐊᖅᓇᙱᔾᔭᐅᑏᑦ ᐱᑕᖅᑲᕈᓐᓇᖅᑐᑦ ᐃᓂᒋᔭᐅᔪᒥᑦ. ᐱᐅᓇᔭᖅᑐᖅ ᐃᓚᐅᑎᑕᐅᒃᐸᑕ ᐋᖅᑭᐅᒪᔪᒥᑦ, ᑭᓯᐊᓂ ᓇᓗᓇᙱᑦᑐᖅ ᑕᒻᒪᖅᓯᒪᓂᖅᑕᖃᓐᓇᖅᓂᐊᖅᒪᑕ. ᖃᐅᔨᒪᔪᒍᑦ ᐊᑐᖅᑕᐅᔪᖅ ᐋᖅᑭᐅᒪᑦᑎᐊᖅᔫᖕᒋᒻᒪ. ᐊᑐᖅᑕᐅᕙᒃᑐᑐᖅᑲᐃᑦ ᑎᖕᒥᓲᒃᑯᑦ ᓈᓴᐃᓃᑦ ᑕᐃᑉᓱᒪᓂ, ᐊᑐᖅᑕᐅᕙᓪᓕᐅᖅᒪᑕ ᑎᑭᐅᑎᓂᖅᑲᓗᐊᖅᓂᐊᖅᒋᓪᓗᑎᒃ ᓇᓚᐅᑦᑖᓇᓂᐊᖅᓗᑎᒃ ᖃᑉᓯᐅᓂᖅᒥᓄᑦ. ᐊᑐᖅᑕᐅᕙᓪᓕᐅᖅᑐᖅᑲᐃᑦ ᐊᑐᖅᑕᐅᖅᑲᑦᑕᖅᑐᑦ ᖁᕝᕙᓯᕈᓯᖅᑐᐃᓂᖅᒧᑦ ᐊᒻᒪᓗ ᓄᓇᓖᑦᑕᐅᖅ ᐅᖃᓚᐅᖅᑐᑦ ᐃᑲᔪᖅᑐᐃᖕᒋᓐᓂᖕᒋᓄᑦ ᖁᕝᕙᓯᕈᓯᖅᑐᐃᓂᖅᒥᒃ, ᐃᓱᒫᓘᑎᖅᑲᖅᑎᑦᑎᐊᖅᑐᑦ ᓇᓄᐃᑦ ᐊᐅᓚᑎᓐᓂᖅ ᐊᔪᖅᑎᑦᑕᐅᖅᑲᑦᑕᖅᓂᖏᓐᓄᑦ. ᑕᐃᒪᓐᓇᒧᑦ ᑎᖕᒥᓲᒃᑯᑦ ᓈᓴᐃᓂᖅ ᓂᕈᐊᖅᑕᐅᓚᐅᖅᑐᖅ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂ, ᑭᓯᐊᓂ ᖃᐅᔨᒪᔪᕐᓇᖅᒥᔭᕗᑦ ᐊᔾᔨᑎᑐᑦ, ᑭᒡᓕᑦᑕᖅᑲᐅᖅᒥᔪᖅ ᐊᑐᖅᑕᐅᓂᖕᒪᑕ.
- **ᓅᓪ:** ᑕᑯᒍᒃᑎᒃ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᒥᓲᑎᒋᔪᑦ, ᑕᑯᒃᓴᐅᑎᑦᑎᔪᖅ ᐊᒥᓲᔪᓐᓃᖅᐸᓪᓕᐊᔪᑦ. ᑭᓇᒥᒃᑭᐊᖅ ᑕᑯᔭᐅᑎᓐᓂᖅᑕᐃᑦ ᑖᓪᓇ, ᐃᓱᒪᒋᔾᔭᖅᑕᖅᑎᑦ ᐅᖃᖅᑕᑎᑦ ᐊᒻᒪᓗ ᑕᑯᑎᑕᑎᑦ ᑭᖑᓪᓕᐅᔪᒥᑦ ᐊᔾᔨᓂᑦ (ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᖃᓄᐃᙵᓂᐅᔪᑦ), ᐊᑐᖅᑕᐃᑦ ᐱᔾᔪᑎᒋᐸᒍ, ᓱᓕᕙ? ᐊᐅᕙᔪᕐᓇᖅᑐᖓ ᐅᖃᕈᖕᒐᓗ, ᑕᒃᑯᒃ, ᐊᒥᓲᔪᕐᓃᓇᐊᖅᑕᖅᑕᕗᑦ ᑲᑎᙳᒥᑦ ᓈᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐊᒻᒪᓗ ᑕᑯᑎᑐᐃᓐᓇᖅᑐᒥᑦ ᐱᖕᓇᖅᐸᓯᖕᒥᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ ᖃᐸᓯᐅᓂᖅᒥᑦ, ᐊᔪᕐᓇᙱᑦᑐᒻᒪᕆᐊᓗᒃ. ᓲᕐᓗ ᑕᐃᒪᓐᓇ ᓴᐳᒻᒥᒃᑯᑦ ᐱᔪᒪᔭᑦ, ᑭᓯᐊᓂ ᓇᒍᓇᙱᑦᑐᖅ ᐋᒃᑲᐅᓂᖅᓯᖅᓂᐊᖅᑕᐃᑦ.
- **ᐊᓕᓴ:** ᐅᖃᐅᑎᔪᓐᓇᖅᑕᒋᑦ, ᐊᓯᖕᒋᖅᓂᒃ ᖃᐅᔨᓴᐃᓯᐊᖅᓯᒪᔪᓐᓄᑦ ᐱᓕᕆᐊᖅᑎᖅᓯᒪᔭᓕᓂᑦ ᐊᒻᒪᓗ ᑖᐸᒃᑯᐊ ᓈᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓴᖅᑭᖅᑕᐅᐸᓗᑎᒃ. ᖃᐅᔨᑎᑦᑎᓂᖅᒧᑦ ᖁᖅᑯᐊᖅᑐᑦ ᓂᐸᓕᖅᑐᑦ. ᖃᐅᔨᒪᔪᑎᑦ, ᐃᓄᐃᑦ ᐅᖃᓪᓚᒃᑐᓂᒃ, ᐆ, ᒪᐃ, ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ ᓱᖁᓕᖅᑐᑦ. ᑕᑯᓯᒪᔪᖓ ᐱᕙᓪᓕᐊᔪᓂᑦ ᐊᒻᒪᓗ ᑭᐅᓚᐅᖅᑕᒃᑲ ᐱᖃᑖ ᐊᔾᔨᐅᑉ ᑕᐅᑐᙳᕆᑕᓯ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖑᑦ ᐅᓂᒃᑳᖅ ᐱᕙᓪᓕᐊᔪᓕᕆᓂᑦ ᐅᔾᔨᕆᔭᐅᖏᓐᓇᑦ. ᐊᖏᖃᑎᒋᑎᑕᖅᑕᕐᒥᑦ.
- **ᓅᓪ:** ᐅᖃᖅᑐᖓ ᐲᖅᓱᒥᑦ ᒪᖅᐳᒃ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ. ᑕᑯᒃᓴᐅᑎᑦᑎᓗᒥᑦ ᐊᑕᐅᓯᕐᒧᑦ ᓇᓂᕐᒥᒃ. ᐊᒃᓯᕈᖅᓱᒥᑦ ᓱᓇᖁᑎᒋᔭᐅᖁᔨᖃᑦᑕᖅᑐᖅ ᓲᖅᓱᖅᑐᑦ ᖃᐅᒃᐸᑦᑕᖅᑲᖏᑦᑐᖅ. ᓂᒋᐊᓂᑦ ᐋᖓᔭᖅᑲᒥᑦ ᑕᑯᔭᐅᑎᓐᓂᐊᓕᖅᐸᐃᑦ ᑭᖖᓂᖓᓄᑦ ᑐᖅᑲᕐᓇᕐᑎᒍᑦ. ᐊᒃᓯᕈᖅᑐᑎᑦ ᑕᒪᑐᒪᖓ ᐱᐅᒋᔪᖅ ᐱᐅᒃᓴᒃᑲᐃᑦᑎᖃᑦᑎᖅᓂᖅᑲᓐᓂᒃ, ᐱᓂᐊᖅᑕᐃᑦ.
- **ᐊᓕᓴ:** ᐅᕙᓂᑦ, ᓱᓕᔪᖅ ᖃᐅᔨᔭᐅᖅᑯᔭᖅ ᑭᑐᑐᐃᓇᓂᑦ ᑐᖅᑲᕐᓇᕐᑎᒍᑦ. ᐊᐅᓚᓐᓂᖅ ᐊᔪᖅᑕᖅ ᑐᓴᒃᓴᐃᑦ ᐊᔪᖅᓇᕐᒪᑦ, ᑭᓯᐊᓂ ᐱᐅᒻᒪᕆᑦᑎᐊᖅᑕᖅ ᐃᓕᒃᑐᑦᑕᐅᖅ ᐅᓂᒃᑳᖑᔪᑦ ᓴᖅᑭᑎᑕᐅᔭᖅᖃᕐᓗᑕ, ᖃᐅᔨᒪᓪᓗᑎ ᓇᓚᐅᑦᑎᓯᒪᙱᓪᓗᑕ.
- **ᑭᓖᑕᓐ:** ᐋᓐᑐᕈ ᑎᑭᑦᓯᒃ ᐊᑐᐊᓂᓚᐅᖅᑕᖅᑎᑦ ᓈᓴᐅᑎᐅᔪᖅ (2021-ᒥᑦ ᐱᖓᓐᓇᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ) ᑐᓴᒃᓴᓂᑦ ᐊᒻᒪᓗ ᖁᕝᕙᓯᕈᓇᐅᑉ ᐊᓂᕝᕙᐊᑕ ᑐᖅᑎᑦᑎᐊᖓᓂᐅᓚᐅᖅᑐᖅ, ᐅᓂᒃᑳᖅ ᓴᖅᑭᖅᑕᐅᑐᐊᕐᒪᑦ. ᑕᒪᓐᓇ ᐱᔾᔪᑕᐅᔪᖅ ᐅᒃᐱᕈᓱᒻᒥᓐᓇᒪ ᓴᐳᒻᒥᒃᑯᓐᓂᑦ ᓴᖅᑭᖅᑕᐅᓚᐅᕐᒪᑦ ᐅᓂᒃᑳᖅ ᓱᕗᓂᐊᒍᑦ ᓴᖅᑭᖅᑕᐅᖕᒪᑦ ᐊᓂᕝᕙᐊᑉ ᑐᖕᖏᓂᑦ. ᐱᔾᔪᑕᐅᔪᖅ ᐱᓕᕆᐊᖑᔪᓇᐊᖅᑲᖅᑐᖅ ᓴᐳᒻᒥᒃᑯᓐᓂᑦ.
- **ᑖᓇ:** ᑐᓂᓯᒍᕕᑦ ᐅᓂᒃᑳᕐᒥᑦ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᖃᓄᐃᓕᓂᖅᑲᖅᑲᑦᑕᖅᐸᑦ ᐅᑯᐊᑑᓐᓃᑦ ᖃᐅᔨᖅᑲᑦᑕᖅᐸᑦ ᑐᖅᑯᖅᑲᓚᐅᕐᒪ? ᖃᐅᔨᐅᓂᖅᑲᓂ ᑎᑭᐅᑎᖅᑲᑦᑕᓂᐅᔪᑦ ᓇᓄᐃᑦ ᐃᓄᖕᓄᑦ, ᓲᕐᓗ ᐅᓗᓇᐊᓇᕐᓂᖅᑲᓂ? ᑕᑯᖅᑲᑦᑕᖅᐸᖅᑲ ᑕᒪᓐᓇ? ᑐᓂᓯᒍᕕᑦ ᐅᓂᒃᑳᕐᒥᑦ, ᖃᓄᒃᑲᓂᖅ ᑕᐃᒪ? ᐊᓯᐊᑕ ᐊᓯᐊᙳᖅᑕᐅᓚᐅᖅᓯᒪᓕᕆᓂᖅᑲᓂ ᓈᓄᒃᑕᐅᔪᖅᑲᖅᑐᑦ ᐅᑯᐊᑑᓐᓃᑦ ᐅᒥᔪᖅᓃᖅᑎᑕᐅᓗᑎᒃ. ᐃᓅᓂᖅᑲᓗᐊᖅᐸᑦ ᖃᐅᔨᔪᖅᑲᓗᒍ, HÅ, ᖃᓄᐃᓕᐅᕈᑎᖅᑲᓐᓂᐊᖅᑲᓪᓚᓐᓂᐊᖅᐸᑕ?
- **ᐊᓕᓴ:** Davis Strait-ᒧᑦ ᑐᓂᐅᑎᒋᓚᐅᖅᑕᐸᑎᖕᓄᑦ, ᑐᓂᓯᓚᐅᖅᑐᒍᑦ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᑲᓂᒃ ᖃᐸᓯᐅᓂᖕᒥᑦ ᐃᖕᒥᖕᓂᒃ ᒥᐊᓂᓇᐸᓗᑎᒃ ᑐᖁᑎᔪᓂᒃ. ᐃᓚᐅᑎᑎᔪᖕᓇᖅᒥᔪᒍᑦ, ᐊᐅᓛᖅᑎᑎᓂᒃᑯᓐᓂᑦ ᐱᔭᓇᐊᖅᑲᖅᑕᖅ, ᐱᑕᖅᑲᑦᑕᓂᖕᒋᓐᓂᑦ

ᐅᓂᑉᑳᑦ ᖃᑉᓯᐅᓂᖅᒋᓐᓄᑦ ᐃᓗᐃᑐᑕᐅᔪᑦ. ᐃᓱᒪᔪᖓ ᐅᖃᐅᓯᐅᔪᖅ ᐱᑦᑕᐅᔫᖓᒪᑦ ᐱᒻᒪᕆᐅᑉᓗᓂ ᐅᕙᑉᑎᖕᓄᑦ ᐃᓚᐅᑎᓐᓂᐊᕐᓗᒍ.

- ᑭᖑᓪᓕᖅᐹᒥᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ (NWMB) ᑲᑎᒪᑎᓪᓗᒋᑦ, ᑐᓴᓚᐅᖅᑕᓪᓂᑦ, ᓄᓇᓕᖕᒥᐅᑕᓂᒃ ᑲᑎᒪᔪᓃᑦᑐᓂᒃ ᖃᐅᔨᑎᑦᑎᓚᐅᖅᑐᑦ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᖅᑭᓪᓗᑎᒃ ᐊᒻᒪᓗ ᐱᔾᔪᑎᖃᓗᐊᓚᐅᖅᑐᖅ ᐃᓱᒪᓕᐊᖑᓂᐊᖅᑐᒧᑦ ᐊᒥᓱᙳᓇᐊᕐᓂᐊᕐᓗᒋᑦ ᑲᑎᓗᒃᑖᕐᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᑕᐃᑉᓱᒪᓂᒥᑦ.
- **ᐊᔾᕈᐊ:** ᐋ, ᑕᐃᒪᓐᓇᐃᓕᓚᐅᖅᑐᖅ ᑕᖅᖀᓐᓇᖅᔪᐊᖅ 2018-ᒥᑦ – ᑭᖑᓪᓕᖅᐹᒥᑦ ᑲᑎᒪᑎᑦᑎᖕᒪᑕ. ᐃᓄᐃᑦ ᓇᓄᕐᓄᑦ ᑐᖁᑕᐅᔪᖅᑲᖅᖄᖅᑎᓐᓇᒎᓚᐅᖅᑐᖅ – ᑭᖑᓂᐊᒍᑦ ᐊᐅᔭᒃᑰᓚᐅᖅᑐᖅ. ᐃᑲᔪᓚᐅᖅᑐᖓ ᑎᑎᕋᖅᑕᐅᑉᓗᒍ ᑎᓂᔭᐅᔪᖅ ᐊᒻᒪᓗ ᑐᓂᑉᓗᒍ ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ. ᐅᖃᐅᓯᕆᓐᓇᖅᑲᑦᑕᖅᑕᕗᑦ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᑦᑕᓇᐊᖅᑲᙱᓐᓂᖅᒋᓐᓄᑦ. ᔫ ᓴᕕᑲᑖᖅ ᓄᑲᖅᖤᖅ ᓈᓴᐅᑎᓂᒃ ᐱᓱᒪᓚᐅᖅᑐᖅ ᖃᑉᔫᖅᑕᖅᓗᑎᒃ ᐅᓂᑉᑳᓕᐅᓚᐅᕐᓂᖅᒋᑦ ᓇᓄᕐᓄᑦ ᐱᓱᒪᔪᓂᒃ, ᐱᓕᕆᓂᖅᒋᓐᓂᑐᐃᓐᓇᖅ. ᑕᒪᓗᒃᑖᖑᖅᒋᑦᑐᖅ, ᑭᑐᓗᒃᑖᖑᖅᒋᑦᑐᑦ ᓇᓄᕐᒥᑦ ᐃᒡᓗᖅᔪᐊᖅᖀᖅᒋᓐᓄᐊᖅᑐᖅᑲᖅᑐᓂᑦ ᖃᓄᑐᐃᓐᓇᖅᔪᐊᖅᓂᑦ. ᐱᓱᒪᓚᐅᖅᑕᕗᑦ ᓈᓴᐅᑏᑦ ᐅᓂᑉᑳᒥᑦ ᐊᒻᒪᓗ, ᐃᓱᒪᔪᖓ, ᐆᒪᔪᓕᕆᔨᕐᔪᐊᓛᒃᑯᓐᓄᑦ ᐃᒃᓯᕙᐅᑕᖅ ᑕᒪᒃᑯᐊᕋᓗᐃᑦ ᖃᐃᑉᓗᑎᒃ ᐊᒻᒪᓗ ᖃᐅᔨᑎᑦᑎᓚᐅᕐᒥᔪᑦ ᑕᒪᒃᑯᓂᙵᑦ. ᓴᙱᑦᑎᐊᓚᐅᖅᑐᖅ ᐱᔾᔪᑕᐅᔪᖅ ᐊᒻᒪᓗ ᓄᓇᕗᒻᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐅᖃᓚᐅᖅᑐᑦ ᐱᒻᒪᕆᐅᓂᖓᑕ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᖅᒋᑦᑐᑎᒃ ᐃᓱᒪᓕᐅᕈᑎᒋᔭᖅᒋᓐᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᖃᐅᓯᖅᖃᖅᓗᑎᒃ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᑦᑕᓇᐊᖅᑲᙱᓐᓂᖅᒋᓐᓂᑦ. ᑕᐃᒪᓐᓇᒧᑦ, ᐅᖃᕈᖕᓇᖅᑐᖓ ᑕᐃᒃᑯᐊ ᓈᓴᐅᑏᑦ ᐱᑐᒋᑦ. ᐆᒪᔪᓕᕆᔨᕐᔪᐊᓛᒃᑯᓪᓘᓐᓃᑦ, ᖃᐅᔨᒪᒍᕙᑦ ᓇᓄᕐᒥᑦ ᑎᑭᐅᑎᔪᖅᑲᓚᐅᖅᐸᑦ, ᐃᒻᒪᖄ ᐊᓕᓚᔪᒧᐊᖅᑐᒋᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓱᓇᒥᒃᑭᐊᖅ ᐅᖃᐅᑎᔭᐅᓯᒪᔪᓂᒃ – ᐃᓄᐃᑦ ᖃᐅᔨᒪᑎᑦᒋᑦ ᖃᓯᑉᓂᒃ ᖃᐅᔨᒪᔭᖅᑲᖅᒐᓘᖅᐱᑦ.

- **ᕼᐃᐅᓇ:** ᐅᑭᐊᒃᓵᖅ ᑖᒃᑖᖅᑎᑦᑎᔭᓇᐊᖅᑲᓚᐅᖅᑐᒍᑦ ᐃᓗᐃᑐᑕᐅᔪᒧᑦ ᓇᓄᕐᒧᑦ. ᐃᒡᓗᖅᔪᐊᕌᖅᑲᖅᑐᑦ ᑕᑯᐃᓐᓇᓕᓚᐅᖅᒪᑕ ᐃᒡᓗᖅᔪᐊᕌᖅᒋᓐᓄᙵᐅᔪᓂᑦ. ᐃᓗᐃᑐᑕᐅᖅᒋᓐᓇᖅᑐᑦ ᐅᑭᐅᑕᒫᒃᑯᑦ, ᐃᓂᑦᑕᐃᓐᓇᕐᒥᑦ. ᓇᓄᐃᑦ ᑕᑯᔭᐅᖃᑦᑕᖅᑐᑦ ᐅᑯᓇᓂ ᕿᑭᖅᑕᓂᑦ, ᐊᐅᔭᒃᑯᑦ ᑕᖅᕿᐅᔪᓂᑦ ᖃᓂᑦᑐᓂᑦ, 5 ᑭᓛᒥᑕᓂᒃ ᖃᓂᑎᒋᔪᓂᒃ ᖃᓂᖕᓂᖅᓴᐅᔪᓂᓘᓐᓃᑦ.
- **ᑖᓇ:** ᓱᑯᔭᐅᖃᑦᑕᖅᑐᒧᑦ ᐃᖕᒥᖕᓂᒃ ᒥᐊᓂᖅᓯᓂᕐᒧᑦ ᑐᖁᑕᐅᔪᓄᑦ. ᑖᒃ ᐱᒋᔭᑉᓯᖕᓄᑦ ᐃᓚᒋᔭᐅᔪᑦ.

- **ᕈᐃᒪᓐ:** ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒥᑦ ᑭᓪᓕᖓᓄᑦ ᓄᖅᑲᖅᕙᓂᖅᑲᑦᑕᖅᑕᓯ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ, ᑕᓯᐅᔭᕐᒧᐊᕈᓐᓇᙱᑦᑐᓯ ᐅᕝᕙᓘᓐᓃᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒧᑦ?
- **ᐊᓕᓴ:** ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖅᒋᓐᓂᒃ ᓈᓴᐃᔭᕚᖓᐸᑕ, ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒥᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒥᑦ ᐊᑕᐅᑦᑎᒃᑰᖅᑲᑦᑕᖅᑕᕗᑦ. ᑭᓯᐊᓂ ᓇᓚᐅᑦᑐᑎᑦ ᐱᓕᕆᐊᕆᖃᑦᑕᙱᓐᓇᑉᑎᒍᑦ ᑕᓯᐅᔭᖅ ᐊᒻᒪᓗ ᐃᓱᒪᔪᖓ ᐊᒃᓱᕈᕐᓇᐅᑕᐅᓗᐊᖅᑐᖅ ᑕᓯᐅᔭᕐᒧᑦ ᐊᖅᒋᔪᒻᒪᕆᐊᒎᖅᒪᑦ ᐊᒻᒪᓗ ᐃᓂᓕᐅᕆᓂᕐᒧᑦ ᐊᒻᒪᓗ ᑮᓇᐅᔭᕐᒧᑦ ᐊᒃᓱᕈᕐᓇᖅᑎᑦᑎᔪᖅ. ᐱᕝᕙᐅᔪᓂᒃ ᐱᑕᖃᖓᖅᒋᑦᑐᒧᑦ ᑕᒪᕐᒥᓗᒃᑖᖅ ᐱᖓᓱᐃᑦ ᐱᓕᕆᐊᕆᓂᐊᖅᑐᒋᑦ, ᑭᓯᐊᓂ ᐃᓱᒪᔪᖓ ᐅᖃᐅᓯᕆᔭᐃᑦ ᐊᐅᓚᓐᓂᖅᒋᓐᓂᖃᕈᓐᓇᖅᑐᑦ ᐊᑯᓐᓂᖅᒋᑦ ᐱᖓᓲᔪᑦ ᐊᒻᒪᓗ ᐄ, ᓇᓗᓇᙱᑦᑐᖅ. ᐃᓱᒪᔪᖓ ᐃᓚᒋᔭᐅᖕᒪᑦ ᑭᓪᓕᖅᑲᖅᓂᐅᔪᓄᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖅᒋᓐᓂᒃ ᓈᓴᐅᑎᓄᑦ. ᑭᓯᐊᓂ ᐅᖃᓚᐅᖅᒪᑦ ᐊᔾᔨᒋᙱᑦᑐᑎᓂᒃ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᑕᑯᓂᐊᖅᑕᕗᑦ ᐊᕙᓐ-ᑐ ᖃᐅᔨᓴᖅᑲᑎᒋᔭᖅᒋᑦᑕᓗ ᐱᓕᕆᐊᕆᖃᑦᑕᖅᑕᖅᒋᓐᓂᒃ, ᐊᒻᒪᓗ ᖃᓄᖅ ᐱᖃᑦᑕᖅᒪᖔᑕ ᐊᐅᓚᓂᕆᔭᐅᔪᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓂᖅᒋᑦ ᐅᖃᐅᓯᔪᑦ ᐊᑯᓐᓂᖅᒋᓐᓂᑦ ᑭᓪᓕᐅᔪᓂᑦ.
- **ᑖᓇ:** ᑭᓪᓕᓂᒃ ᐅᖃᐅᓯᖅᖃᖅᑎᓪᓗᑕ, ᑕᑯᔪᖕᓇᖅᑐᓯ ᐅᕙᓂ (ᑕᑯᑎᑦᑎᔪᖅ ᓄᓇᙳᐊᕐᒥᑦ ᓴᓂᕋᕐᒦᑦᑐᒥᑦ) ᐃᓄᐃᑦ ᐱᖅᑯᓯᓕᕆᔨᒃᑯᓐᓂᙵᖅᑐᒥᒃ. ᐃᓂᐅᔪᑦ ᐊᑎᖅᒋᓐᓂᑦ ᐱᓕᕆᔪᑦ. ᐊᑕᐅᓯᖅ ᒥᑭᔫᔪᖅ ᐃᓂᐅᔪᖅ ᐊᒻᒪᓗ ᑕᐃᒪᓐᓇ ᐊᒥᓲᑎᒋᔪᑦ ᐊᑏᑦ ᐊᒻᒪᓗ ᐊᖅᒋᔫᑉᓗᑎᒃ ᖃᑕᙳᑎᒌᙳᓚᐅᖅᑐᑦ ᑕᐃᑉᓱᒪᓂᒥᑦ ᓇᒻᒥᓂᖅ ᓄᓇᖃᖅᐸᓚᐅᖅᑐᑦ. ᖃᑕᙳᑎᒌᒍᑦ ᓄᓇᐃᑦ ᐱᒋᔭᐅᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ ᑕᐃᒪᓐᓇ ᐊᒥᓲᑎᒋᔪᑦ ᐊᑏᑦ ᐊᖅᒋᑎᒋᔪᒥᒃ ᐃᓐᓯᒋᔭᐅᔪᓗᒃᑖᔭᔪᓂᑦ [illegible]

- **ᑯᓇᔅ:** ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖅᒋᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᓄᓇᙳᐊᖅ, ᐱᑕᖃᖅᐱᓯ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒥᑦ?
- **ᐊᓕᓴ:** ᐱᑕᙱᑦᑐᖅᒪ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒧᑦ. ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᒪᒋᓚᐅᙱᑦᑕᖓᑦ ᓈᓴᐃᓂᖅ. ᑲᒪᒋᔭᐅᓚᐅᖅᑐᑦ ᐋᓐᑎᐅᕆᐅᒥᑦ ᒐᕙᒪᒃᑯᓐᓂᑦ ᐊᒻᒪᓗ ᐅᓂᑉᑳᖓᑦ ᐱᓕᕆᖃᑦᑕᖓᖅᑐᑦ.
- **ᑯᓇᔅ:** ᐃᓚᖅᒥᑦ ᓇᓄᐃᑦ ᑭᓪᓕᐅᔪᑦ ᐅᖔᓇᙵᐅᓂᐊᖅᑐᑦ. ᑕᐃᒪᓐᓇᒧᑦ ᓇᓚᐅᑦᑎᓯᒪᙱᑦᑐᖅ.

- **ᐊᓕᓴ**: ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᑲᓂᒃ ᐱᖃᙱᑦᑐᒍᑦ ᑕᓯᐅᔭᕐᒥᑦ ᐅᐊᖕᓇᖓᓄᑦ ᓈᓴᐃᓂᐊᕋᑉᑕ ᐊᐅᔭᐅᓂᐊᖅᑐᒥᑦ. ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᑲᓂᒃ ᐱᓯᒪᓂᐊᖅᑐᒍᑦ ᐊᐅᔭᕐᒍᑦ, ᑭᓯᐊᓂ ᓄᓇᙱᐊᓕᐅᕐᓂᐊᖅᑐᒍᑦ ᐊᕝᕙᕐᓚᕆᒋᑐᒍ ᑕᓯᐅᔭᕐᒍᑦ ᐊᑐᖅᑐᓂᑦ.

ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔩᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔩᒃᑯᑦ (ECCC) ᖃᐅᔨᓴᐃᓂᖓᑦ

- **ᓵᒻ**: ᖃᑉᓯᓂᒃ ᓇᓄᕐᓂᑦ ᐆᒃᑐᕋᐅᑎᓂᒃ ᐱᓯᒪᕙᓯ?
- **ᐊᕙᓐ**: ᓂᕿᖏᓐᓂᒃ ᐱᓯᒪᔪᓄᑦ, ᐃᓱᒪᔪᖓ 900 ᐅᖓᑖᓄᑦ ᐆᒃᑐᕋᐅᑎᖃᖅᑐᑦ ᐊᑐᓂ ᓇᓄᕐᓄᑦ. ᐊᑯᓐᓂᖓᓃᑦᑐᖅ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸ ᐊᒻᒪᓗ ᓂᒥᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸ.
- **ᓵᒻ**: ᐊᑕᐅᓰᖅᓱᒥᑦ ᐆᒃᑐᕋᖅᓯᒪᔭᓯ ᐅᕝᕙᓘᕝᕙ ᐅᑭᐅᑕᒫᑦ?
- **ᐊᕙᓐ**: ᖃᑉᓰᖅᑕᖅᓱᑕᐅᓕᖅᑐᖅ. ᐃᓛᓐᓂᒃᑯᑦ ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᐊᑕᐅᓰᖅᑕᐅᑐᐃᓐᓇᖅᓱᑎᒃ ᑕᑯᔭᐅᓯᒪᔪᑦ. ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᒪᕐᕈᐃᖅᑕᐅᑉᓗᑎᒃ ᐅᕝᕙᓘᓐᓃᑦ ᐱᖓᓱᐃᖅᓱᒥᑦ. ᐊᔾᔩᙱᖃᑦᑕᖅᑐᖅ. ᑭᓯᐊᓂ ᑎᑎᕋᖅᐸᓪᓕᐊᓯᒪᔪᑦ ᖃᐅᔨᒪᔭᐅᔪᑦ ᓇᓄᕐᓂᑦ, ᑐᑭᓯᐅᒪᔾᔪᑎᒃᑲᓐᓂᕐᓂᖅ ᐱᖃᑦᑕᖅᑐᑦ ᑕᐃᒪᓐᓇᐃᓕᐅᕆᐊᖅᑐᕐᓂᑕᒫᑦ.

- **ᑭᓵᑕᓐ**: 2021-ᒥᑦ ᐊᓄᕞᖃᑦᑕᖅᑐᒻᒪᕆᐅᓚᐅᖅᑐᖅ. ᐅᑉᓛᒃᑯᑦ ᑐᐸᓚᐅᖅᓯᒪᔪᖓ Hᐃᐅᓇ ᑲᓱᒃᑐᖅᑐᖅ ᐅᖃᐅᑎᑉᓗᓂᖓ ᑎᖕᒥᓲᕋᒎᖅ ᐅᖁᓯᒪᔪᖅ.

- **ᐊᔾᕿᐊ**: ᐊᒥᓲᓂᖅᓴᑲᓪᓚᖕᓂᑦ ᓂᕿᖏᓐᓂᑦ ᐆᒃᑐᕋᐅᑕᐅᓂᐊᖅᑐᓂᑦ ᐱᓕᕆᖅᑲᑦᑕᖅᓯᒪᔪᑦ ᐱᖓᖕᓇᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸ ᓂᒥᐊᖅᐸᓯᐊᓂᐅᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸ. ᐊᒥᓱᐃᑦ ᓇᓄᐃᑦ ᐆᒃᑐᕋᐅᑕᐅᓂᐊᖅᑐᓂᒃ ᐱᕖᐅᓚᐅᕋᐅᔭᖅᑎᑦᑐᒥᑦ ᐱᖓᖕᓇᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸᒥᑦ, ᐱᓕᕆᓂᖅᑕᖅᑲᖏᓐᓂᖅᓴᐅᓂᖓᓄᑦ ᓂᒥᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸᒥᑦ, ᓇᓄᐃᑦ ᐅᑎᖅᑕᖅᑲᑦᑕᕈᓐᓇᖅᑐᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᖃᓄᐃᓕᐅᕐᓂᖏᑦ ᐱᔭᐅᓯᒪᙱᑦᑐᑦ ᐊᒻᒪᓗ ᖃᓪᓚᒃᑯᓚᐅᕐᓂᖓᑕ ᖃᐅᔨᓴᐃᓂᐅᑉ.
- **ᐊᕙᓐ**: ᐃ, 100%. ᓇᓗᓇᙱᑦᑐᖅ ᐊᑯᓂᐅᓂᖅᓴᒥᒃ ᖃᐅᔨᓴᐃᓂᖅᑕᖅᑲᑦᑕᖅᓯᒪᔪᖅ ᐱᖓᖕᓇᖅᐸᓯᖓᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓂᒋᖅᐸᓯᐊᓂᐅᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᐸ, ᑭᓯᐊᓂ ᐱᓕᕆᐊᖅᑲᖅᓯᐃᓐᓇᖅᑐᑦ ᑕᒪᐃᓐᓂᑦ ᑭᓪᓕᒥᔭᐅᔪᓂᑦ ᐱᖓᓱᑦ ᐅᑭᐅᓅᓕᖅᑐᖅ, ᑕᐃᒪᓐᓇᓗᑦ ᐃᓱᒪᔪᒧᑦ ᑕᑯᔭᐅᔪᖕᓇᖅᑐᑦ ᐱᕙᓪᓕᐊᓕᖅᑐᑦ, ᐆᒃᑐᕋᐅᑕᐅᓂᐊᖅᑐᑦ ᐊᖏᑎᒋᓂᖏᓐᓂᒃ ᓈᒻᒪᒃᑑᔪᓂᒃ ᐱᖃᑦᑕᓕᖅᑐᒧᑦ.

- **Hᐃᐅᓇ**: ᖃᐅᔨᒪᕙᒥᑦ ᐅᑎᖅᑕᕐᓂᖕᒥᑦ? ᖃᐅᔨᒪᕙᑦ ᓇᒧᙵᐅᖃᑦᑕᕐᒪᖔᑕ ᐅᖅᑳᖕᖓᕙᑦ ᓅᑲᑦᑕᖅᑲᑦᑕᕐᒪᑕ?
- **ᐊᕙᓐ**: ᑐᑭᓯᐊᓂᓇᓴᕋ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒥᑦ ᓈᓴᐅᑎᓂᑦ/ᑎᑎᖅᑲᓂᐅᔪᖅ ᐊᒻᒪᓗ ᖁᖓᓯᕈᑎᖏᑦᑎᒍᑦ ᐊᐅᓚᓂᖏᓐᓂᑦ, ᑭᓯᐊᓂ ᐃᓛᓐᓂᒃᑯᑦ ᓇᓄᐃᑦ ᒥᑭᓂᖅᓴᓂᑦ ᐃᓂᐅᔪᓐᑐᒪᖃᑦᑕᖅᑐᑦ. ᐊᓯᖏᑦ ᓇᓄᐃᑦ ᐊᖏᓂᖅᓴᒥᑦ ᐃᓂᖃᖅᑐᑦ ᐊᒻᒪᓗ ᐊᓯᖏᑦ ᓇᓄᐃᑦ ᐊᐅᓪᓚᕆᐊᖅᓱᑎᒃ ᐊᒻᒪᓗ ᐱᓱᒃᐸᒃᓱᑎᒃ ᑲᓇᑕᐅᑉ ᐅᐊᖕᓇᖓᑕ ᐃᓱᐊᓄᑦ ᐊᒻᒪᓗ ᐊᑯᑭᑦᑐᒧᐊᖅᓱᑎᒃ. ᐊᒥᓱᓂᒃ ᐊᕝᕌᒍᓂᑦᑐᓂᒃ ᐱᓕᕆᖃᑦᑕᖅᑐᑦ.
- **Hᐃᐅᓇ**: ᐅᑭᐊᒃᓵᓚᑕᒫᑦ ᑕᐃᒪᓐᓇᐃᖅᑲᑦᑕᖅᑐᖅ, ᓂᓇᐅᒥᔅᓯᑎᒍᑦ ᓇᓄᐃᑦ ᐅᑎᖅᑕᕐᓇᖕᖓᓗᑦ ᐅᔮᓇ, ᐅᑎᖅᑕᖅᑐᑦ ᐅᐊᖕᓇᒧᑦ ᓯᑯᑕᖅᑲᖅᑕᖅᓯᓂᖓᖕᓂᖕᖓᓗᐊᑦ ᐊᒻᒪᓗ ᓇᑎᖅᑕᖅᑐᖅᓯᓂᖓᖕᓂᖕᖓᓗᐊᑦ. ᐅᑭᐅᑕᒫᑦ ᐊᕝᕌᒍᓇᖅᑲᑦᑕᖅᑑᒃ. ᓯᐅᔭᒃᑯᓪᓗᓂ ᐅᑎᖅᑕᖅᑲᑦᑕᖅᑐᑦ. ᐅᑎᖅᑐᐃᓐᓇᖅᑲᑦᑕᖅᑐᑦ ᐅᖃᓪᓚᒃᑑᖃᑦᑕᖅᑐᑦ. ᖃᐅᔨᒪᓂᖃᖅᑎᓐᓂᐊᖅᐸᓯ ᓇᑭᑦ ᐱᒋᐊᖅᑕᖅᑎᓯᒪᕙᑦ?
- **ᐊᕙᓐ**: ᑰᓪᔪᐊᒃᑰᖅᑲᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᐊᖅᕙᐊᑦ ᒥᒃᓴᓄᙵᐅᑉᓗᑎᒃ ᐅᑭᐊᒃᓴᒃᑯᑦ. ᖁᖓᓯᕈᑎᓂᑦ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᑲᓂᒃ ᐱᖅᑐᑦ ᑕᑯᑎᓐᓂᐊᖅᑕᕗᑦ ᐊᐅᓚᓂᖏᓐᓂᑦ.

20:53-20:59 ᓄᖅᑲᖓᓚᐅᐱᓪᓚᖕᓂᖅ

- **ᑭᓵᑕᓐ**: ᖃᑯᒎ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᓐᓄᐊᖅᑕᐅᓂᐊᖅᐸᑦ?
- **ᐊᓕᓴ**: ᑭᖑᓪᓕᕐᒥᐅᓂᐊᖅᑐᖅ ᒪᓐᓃᑦ-ᒥᑦ, ᑭᓯᐊᓂ ᓱᓕᑦᑯᑦ, ᖃᓄᑎᒋᐅᓂᖓ ᐱᓕᕆᓂᕆᔭᐅᔫᑉ ᑐᓂᓯᓂᐊᕐᓗᓂ, ᑐᓂᓯᕙᖕᓇᕐᓂᐊᖅᑐᕆᑦᑐᑦ ᑲᑎᒪᓂᕐᒧᑦ. ᐅᑭᐊᒃᓵᒃᑯᑦ ᑲᑎᒪᓂᕐᒥᐅᓂᐊᖅᑐᖓᐅᔪᖅ.
- **ᔭᐋᓐ**: ᓴᒡᓚᐆᑦ 1 ᓄᑖᖑᔪᖅ ᐅᑭᐅᖅ, ᑲᑎᓗᒃᑖᖅᓱᒥᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐊᓯᐊᙳᖅᑎᔾᔭᖏᑕᕗᑦ ᐅᑭᐅᖑᔪᖅ. ᖃᐅᔪᒥᑦ ᐅᑭᐅᕐᒥᐅᓂᐊᖅᑐᖅ.
- **ᓵᒻ**: ᒐᕙᒪᐃᑦ ᖃᐃᖅᑲᑦᑕᖅᑐᑦ ᐊᑭᓴᖅᑐᖅᓯᓂᐊᖅᓗᓂ ᐃᓱᒪᓕᐊᓇᐊᓂᒃᑕᖕᒥᑦ. 4.5%-ᒦᒃᑯᐸᑕ, ᓲᖅ ᑐᓂᙱᑉᐱᑎᒍᑦ 10%-ᒥᑦ? ᖃᓄᖅ ᐊᓯᐊᙳᖅᓯᓂᐊᖅᐸᑦ ᐱᖁᔭᓂᓇᔭᑉᑎᖕᓂᑦ? ᑲᑎᒪᖃᑦᑕᖅᑐᒍᑦ ᐅᑭᐅᑕᒫᑦ ᐊᒻᒪᓗ ᒪᕐᕉᒃ ᐃᑲᕐᕋᓂᑦ ᑲᑎᒪᓂᕗᑦ ᐊᑦᑐᖅᓯᓂᖃᙱᑦᑐᖅ ᐱᓕᕆᐊᑦ ᐊᑭᓴᖅᑐᖅᑕᖅᑲᓯᓚᑕ ᐃᓱᒪᓕᐊᓇᓚᐅᖅᑕᖅᑲᖅᑐᒋᓐᓂᑦ. ᖃᖓᒃᑯᖓᓂᖅ, ᐆᒪᔪᓕᕆᐊᒃᑯᑦ, ᑖᒃ ᐱᔭᐅᕗᖅᓇᖅᓯᖃᑦᑕᖅᑐᑦ ᐅᑭᐅᓖᑦ 1-ᒥᑦ ᐊᒻᒪᓗ ᑭᖑᓪᓕᖅᐹᖑᔪᒥᑦ ᐃᑲᕐᕋᒥᐅᖃᑦᑕᖅᑐᖅ ᐊᒻᒪᓗ ᐃᓱᒪᓕᐅᕆᐊᖅᑲᖅᑕᖅᑐᒍᑦ. ᖃᐃᐸᑦ ᒪᓕᖅᓂᕆᔭᑉᑎᖕᓂᖅ, ᒥᑭᔪᒥᒃ ᑭᓪᓕᖅᓴᖅᑎᑦᑎᓇᖃᕐᓯᐅᑦᑐᑦ ᐅᖓᑎᓗᐊᖅᐅᕐᓇᓂᑦ ᓇᓐᓄᒃᑎᑦᑎᓂᐊᖅᐅᓐᓇᐸᑕ. ᐊᒻᒪᓗ ᓇᓄᕐᓯᐅᖅᓴᑉᐳᓚᐅᖅᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᑦ ᓴᓇᕙᐃᓯᓇᓚᐅᖅᑲᖅᑐᒍᑦ. ᖃᓄᐃᓕᐅᖅᐸᑦ ᑖᕙᓂ? ᐃᓱᒪᓕᐅᖃᑎᒌᖕᓂᐊᖅᑐᑦ. ᖃᐃᑉᐳᑎ ᐅᖃᐅᑎᑉᐳᑕᑐ ᐱᔨᑕᐅᔪᓂᒃ ᐃᓱᒪᓕᐅᕆᐊᖅᓯᓂ. ᐃᓱᒪᒋᓴᑉᐳᑎᖅᓯᓂᑦ ᑐᓂᓯᑉᐳᑕ. ᖃᓄᐃᑦᑐᒥᒃ ᐊᑐᖅᓯᓂᖅᑎᑦᑎᓂᐊᖅᐸᑦ

ᑲᑎᒪᓂᖅ? ᑕᒪᒃᑯᐊ ᑲᑎᒪᓂᕆᖃᑦᑕᖅᑕᕗᑦ ᐅᑭᐅᑦ 50 ᐅᖓᑖᓄᑦ. ᒐᕙᒪᒃᑯᑦ ᐋᖅᑭᒃᓯᓯᒪᓕᐊᓂᒃᑐᑦ ᐊᒻᒪᓗ ᑲᙳᔫᑎᒋᙱᑕᖓᑦ – ᓄᓇᕗᐊᕐᒥᑦ ᐊᖏᖃᑎᒌᒍᑏᑦ.

- **ᑮᓪᓚᓐ**: ᓇᓄᐃᑦ ᐱᖓᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕝᕙᕐᒥᑦ/ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕝᕙᕐᒥᑦ ᓇᓐᓄᒃᑕᕆᔭᐅᓚᐅᖅᓯᒪᕙᑦ ᑕᓯᐅᔭᕐᒥᑦ?
- **ᐊᓕᓴ**: ᐃᓚᒋᔭᐅᑉᓗᓂ ᑕᓯᐅᔭᕐᒥᑦ ᓈᓴᐃᓂᕆᓂᐊᖅᑕᑉᑎᖕᓄᑦ, ᑐᓂᓯᕙᒪᔪᒍᑦ ᐆᒃᑐᕋᕈᑎᓂᒃ, ᓇᓐᓄᒃᑕᐅᔪᓂᒃ ᐆᒃᑐᕋᖅᑕᐅᓂᐊᖅᑐᓂᒃ ᐊᔾᔨᒋᖓᓄᑦ ᖃᕆᑕᐅᔭᒃᑯᑦ ᑐᖅᑯᖅᓯᓯᒪᙱᖓᒍᑦ ᑖᑉᑯᐊ (ᐱᖓᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕝᕙᕐᒥᑦ/ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕝᕙᕐᒥᑦ) ᐱᔭᐅᓯᒪᔪᓂᑦ, ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᑕ ᐊᔾᔨᒌᒃᑐᖃᕋᙱᑦ ᑖᑉᑯᓇᙵᑦ ᒪᕈᕝᔪᓂᑦ.

- **ᓅᒻ**: ᑕᒪᕐᒥᓗᒃᑖᖅᖃᔅᖅ ᓇᓄᐃᑦ (ᖁᖓᓯᕈᑎᓂᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ) ᑲᖏᖅᓵᓂᕐᒧᙵᐅᓱᐃᑦᑐᑦ. ᖃᐅᔨᒪᒻᒪᓪᓚᑦᑖᖅᐱᑦ ᓇᓄᖅᓱᐅᖃᑦᑕᖅᑕᕗᑦ ᓇᓅᖏᒃᑲᓗᐊᕐᒪᙱᑕ ᑕᓯᐅᔭᕐᒥᑦ?

- **ᐊᔾᕈᐊ**: ᐊᕐᓇᓪᓗᐃᓐᓇᐅᔪᑦ ᓇᓄᐃᑦ (ᖁᖓᓯᕈᑎᓂᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ)?
- **ᐊᕙᓐ**: ᐄ.

- **ᑯᓇᒃ**: ᖃᐸᓯᓂᒃ ᓇᓄᕐᓂᒃ ᖁᖓᓯᕈᓯᖅᓯᒪᕕᓯ ᐱᖓᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕝᕙᕐᒥᑦ?
- **ᐊᕙᓐ**: ᐊᑐᕐᓗᓂ ᓵᐯᐊᔅᖃᖅᑐᒥᒃ ᖁᖓᓯᕈᑎᒥᑦ ᖃᖓᒃᑯᑐᐃᓐᓇᖅ, ᖁᓕᐅᖃᑦᑕᖅᑐᑦ ᖁᖓᓯᕈᑏᑦ ᐃᓕᔭᐅᔪᑦ ᐅᑭᐅᑉ ᐃᓗᐊᓂᑦ, ᐊᒻᒪᓗ ᒪᕐᕉᒃ ᐅᑭᐅᑦ ᖃᓂᑖᓂᑦ ᐊᔪᙱᖃᑦᑕᖅᑐᑦ. ᓇᓕᑐᐃᓐᓇᖏᓐᓂᑦ ᖁᓕᓂᑦ ᐊᕙᑎᓄᑦ ᓇᓄᕐᓄᑦ ᑲᑎᔾᔪᒋᑦ.
- **ᔪᐋᓐ**: ᓯᐅᑎᖏᓐᓄᑦ ᐃᓕᓯᖃᑦᑕᖅᐱᓯ ᐊᖑᔅᓵᓗᓗᒃᑖᓄᑦ?
- **ᐊᕙᓐ**: ᓯᐅᑎᖏᓐᓄᐊᖅᑎᓇᙱᑦᑐᒍᑦ ᐊᖑᔅᓵᓗᕐᓄᑦ. ᓯᐅᑎᖏᓐᓄᐊᖅᑎᓇᖃᑦᑕᖅᖁᖅᑐᑦ ᐃᓗᐃᓗᑕᐅᔪᑦ ᓇᓄᐃᑦ ᓴᐸᑯᑕᐅᔪᓄᑦ.
- **ᐊᓕᓴ**: ᖁᖓᓯᕈᑎᑐᐃᓐᓇᐃᑦ, ᓯᐅᑎᖏᓐᓄᐊᖅᑕᐅᓛᖑᙱᑦᑐᑦ ᑕᕐᓇᔭᐅᑎᒦᑦᑐᑦ. ᓯᐅᑎᖏᓐᓄᐊᖅᑕᐅᓛᓂᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ ᐱᒋᐊᖅᖁᓚᐅᖅᑐᖅ 2017-ᒥᑦ, ᖃᐅᔨᒋᐊᕈᖓᖅᑐᒍᑦ ᓈᓴᐅᑎᐅᔪᑦ/ᑎᑎᖅᖃᐃᑦ ᐊᐅᓚᔪᒧᐊᖃᑕᐅᔪᖓᕐᒪᙱᐸᑎᒍᑦ, ᐊᔾᔨᙱᒋᓐᓂᖏᑦ ᐊᖑᔅᓵᓗᐃᓪᓗ ᐊᕐᓇᓪᓗᐃᓪᓗ ᑕᑯᓂᐊᕐᓗᒋᑦ.

- **Hᐃᐅᓇ**: ᓇᓂ ᓇᓄᐃᑦ ᖁᖓᓯᕈᓯᖅᑐᖅᑕᐅᖃᑦᑕᖅᐸᑦ?
- **ᐊᕙᓐ**: ᐅᐊᐳᒃ ᒥᕐᖑᐃᖅᓯᕝᕕᖏᐅᒐᔪᒃᑐᖅ ᖁᖓᓯᕈᓯᖅᑐᖅᑕᐅᓲᑦ.
- **Hᐃᐅᓇ**: ᓄᓇᕗᑦ ᓯᓪᓭᖓᓃᑦᑐᓂᒃ ᓇᓄᕐᓂᒃ ᖁᖓᓯᕈᖅᑐᐃᓚᐅᖏᑦᑐᑦ?
- **ᐊᓕᓴ**: ᐋᒃᑲ.
- **Hᐃᐅᓇ**: ᐊᒻᒪᓗ ᑕᐃᑉᑯᐊ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓄᑦ?
- **ᐊᓕᓴ**: ᖁᖓᓯᕈᑎᓂᙳᖅᑐᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ? ᐋᒃᑲ.
- **Hᐃᐅᓇ**: ᓲᖅ ᑕᒡᕙ ᐅᕙᐸᑎᖕᓄᑦ ᑕᑯᑎᑕᐅᕙᑦ? ᓲᖅ ᑕᑯᑎᑕᐅᓕᑦ? ᐅᕙᐸᑎᖕᓄᑦ ᐊᑑᑎᖃᕐᓂᐊᖏᑉᐸᑕ?
- **ᐊᓕᓴ**: ᐆᒪᔪᓕᓇᔨᐊᑖᖑᔪᓂᑦ ᑕᑯᔪᒪᐸᓗᑎᒃ ᖁᖓᓯᕈᑎᓂᙱᖅᑐᓂᒃ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᖃᓂᒃ ᖃᐅᔨᑎᑕᐅᓚᐅᖅᓯᒪᔪᒍᑦ, ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ ᓱᓇᒧᑦ ᐊᑐᖅᑕᐅᒪᙱᑕ. ᐅᖃᐅᓯᓇᔭᖏᑦ ᒪᓕᒃᓱᒋᑦ, ᐋᖅᑭᒃᓱᓚᐅᖅᑕᒃ ᐊᐅᓚᔪᐃᑦ ᑕᑯᑎᑦᑐᖓᕐᓂᐊᕋᐸᑦ ᐆᒪᔪᓕᓇᔨᐊᑏᖑᔪᑦ.

ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᖏᓪᓗᑎᒃ ᐱᓕᕆᐊᒃᓴᐃᑦ ᐱᖓᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕝᕙᕐᒥᑦ

- **Hᐃᐅᓇ**: ᑐᖑᔭᖅᑐᖅ ᒥᖑᐊᖅᑕᐅᓯᒪᔭᖅ ᑕᐃᑲᓃᖅᑕᑦᑕᖅᑐᖅ ᓂᕿᖓᓂᒃ ᐲᖅᓯᕝᕕᐅᖅᖄᖅᑎᓪᓗᒋᑦ, ᐲᖅᑕᑦᑕᖅᐸ? ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᔪᒥᑦ?
- **ᐊᕙᓐ**: ᓲᕐᓗ ᐃᒫᓃᓚᐅᖅᓱᑎᒃ ᐲᖅᑕᑦᑕᖅᑐᖅ. ᐄ, ᐃᒪᕐᒧᑦ ᐲᕈᖓᓇᖅᑐᖅ. ᐲᖅᑕᑦᑕᖅᑐᖅ, ᓲᕐᓗ, ᐱᓇᓱᐊᕈᓯᖅ ᐱᐊᓂᒃᐸᑦ.
- **Hᐃᐅᓇ**: ᑕᐃᒪᓕ ᐅᖃᐅᓯᓇᖅᖃᐅᔭᐃᑦ ᐊᑐᓗᐊᖏᑦᑐᖅ, ᐅᖃᖅᖃᐅᒐᕙᑦ ᓇᓄᖅ ᒪᕿᐃᖅᑕᕐᓗᒍ ᐱᓂᐊᖏᑕᐃᑦ, ᑭᓯᐊᓂ ᐃᒫᓃᕋᓂᑐᐊᖅᐸᑦ…
- **ᐊᕙᓐ**: ᐱᓕᓇᖃᑦᑕᖅᑐᒍᑦ ᓯᓪᓭᑎᑰᖅᓱᑕ, 100 ᑮᓚᒥᑕᓂᒃ ᓯᓪᓭᒃᑰᖃᑦᑕᖅᑐᒍᑦ, ᑲᔪᓯᒥᐊᕐᓗᑕ. ᑕᐃᒪᓐᓇᒧᑦ ᐊᑕᐅᓯᑦᑕᐃᓐᓇᕐᒥᑦ ᐆᒃᑐᕋᕈᑎᐅᓂᐊᖅᑐᓂᒃ ᐱᖃᑦᑕᙱᑦᑐᒍᑦ.
- **ᐊᓕᓴ**: ᒥᖑᑎ ᐊᑐᖅᑕᐅᑎᓪᓗᒍ ᓇᓄᕐᓂᑦ ᓄᓇᓕᖕᒦᑦᑐᓄᑦ, ᒥᖑᑎ ᐊᑐᖅᑕᐅᔪᓇᓗᐊᕐᓂᐊᙱᑦᑐᖅ, ᐅᖃᕋᕙᑦ Hᐃᐅᓇ.

- **ᑖᓂᐅᓪ**: ᐅᑭᐅᑕᒫᑦ ᓇᓐᓄᒃᑕᐅᔪᓂᑦ ᐆᒃᑐᕋᐅᑏᑦ ᐊᐅᓪᓚᖅᑎᖃᑦᑕᖅᑕᕗᑦ. ᓱᔭᐅᕙᑦ ᑕᐃᑯᐊ? ᐊᑐᖅᑕᐅᖃᑦᑕᖅᐸᑦ ᓂᕿᖏᓐᓂᑦ ᐲᖅᑕᐅᔪᓂᒃ ᐆᒃᑐᕋᐅᑎᓄᑦ?
- **ᐊᓕᓴ**: ᐄ, ᐊᕙᓐ ᑕᑯᑎᖅᑲᐅᔭᖓ ᑕᒪᓇ, ᓇᓐᓄᒃᑕᐅᓯᒪᔪᑦ, ᓄᓇᕗᒻᒧᑦ ᓈᓴᐅᑎᐅᖅᑲᐅᔪᑦ/ᑎᑎᖅᑲᐅᏄᑦ. ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᓐᓃᖅᑕᐅᖅᑲᐅᔪᑦ ᓂᕿᖏᓐᓂᑦ ᐲᖅᑕᐅᔪᓂᑦ ᐆᒃᑐᕋᐅᑎᓄᑦ. ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓂᑦ, ᐱᓯᒪᔭᕗᑦ ᐆᒃᑐᕋᐅᑕᐅᔪᓗᒃᑖᑦ ᐊᒻᒪᓗ ᐱᕕᖃᓗᐊᖏᑦᑐᑦ ᖃᐅᔨᓴᕐᓂᐊᕐᓗᒋᑦ ᐊᒥᓲᑦ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ. ᐊᒥᓱᑦ

ᓈᓴᐅᑏᑦ/ᑎᑎᖅᑲᐃᑦ ᑐᒃᓯᕋᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᐱᓕᕆᐊᖃᖅᑐᓄᑦ ᓇᑭᙵᕆᒪᙵᑕ ᖃᐅᔨᓴᐃᓕᖅᓱᑎᒃ. ᐅᖃᓚᐅᕋᒪᑦ ᑕᓯᐅᔭᕐᒥᑦ ᓈᓴᐃᓂᕐᒧᑦ ᐸᕐᓇᒃᑐᒍᑦ ᐊᑯᓪᓕᕈᕐᕕᖕᒧᑦ, ᓂᕿᖏᓐᓂᒃ ᐆᒃᑐᕋᐅᑎᓂᒃ ᐱᖃᑦᑕᕈᒪᔪᒍᑦ ᓈᓴᐃᑎᓪᓗᑕ ᑕᐃᑲᓂ. ᑲᑎᑦᑎᓂᐊᖅᑐᒍᑦ ᐃᓚᖏᓐᓂᒃ ᐆᒃᑐᕋᐅᑕᐅᓂᐊᖅᑐᓂᒃ ᑕᓯᐅᔭᕐᒥᑦ ᓄᓇᕐᓂᑦ. ᐃᓚᒋᔭᐅᔪᖅ ᑕᒪᑐᒎᒐ ᑐᓂᓯᔪᒪᖕᒥᔪᒍᑦ ᐃᓚᖏᓐᓂᑦ ᓇᓄᒃᑕᐅᓯᒪᔪᓂᒃ ᐆᒃᑐᕋᐅᑎᓂᒃ ᑕᓯᐅᔭᕐᒥᑦ ᐆᒃᑐᕈᒪᐸᓗᒋᑦ ᐊᒻᒪᓗ ᖃᐅᔮᓂᐊᕐᓗᒋᑦ. ᐃᓱᒪᔪᖕᒐ ᐋᑑᑎᖃᖅᑑᓂᐊᖅᑐᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᐱᔭᕗᑦ ᑖᙳᖓᑦ ᐱᔪᒪᓂᖅᑕᖃᒪᑦ ᐊᒻᒪᓗ ᑲᖏᖅᕼᓂᕐᒥᐅᑕᐃᑦ ᓇᓄᖃᑦᑕᕈᖕᓇᕐᒪᑕ ᑕᓯᐅᔭᕐᒥᑦ ᓇᓄᖏᓐᓂᑦ. ᐱᒻᒪᕆᐅᓂᐊᖅᖠᖅᑐᖅ ᐃᓱᒪᔪᖕᒐ ᑕᓯᐅᔭᕐᒥᑦ ᓈᓴᐃᓂᖅ ᑐᓂᐅᑎᒋᓂᐊᕐᓗᒍ ᐆᒃᑐᕋᐅᑕᐅᓚᐅᖅᑐᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᓐᓈᕐᓂᐊᕐᓗᒋᑦ ᐱᙳᙵᖅᐸᓯᙵᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ/ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ, ᑕᐃᒪᓐᓇ ᑐᑭᓯᐊᓂᖅᓴᐅᓕᕐᓂᐊᕋᐸᑕ ᑕᒪᕐᒥᓪᓗᒃᑖᖅ ᑕᓯᐅᔭᕐᔪᐊᕐᒦᑦᑐᓂᒃ, ᓇᑭᙵᖅᓯᒪᓂᖏᓐᓂᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓐᓂᕆᔭᖏᓐᓂᑦ.

- **ᔪᐋᓐ**: ᐊᐱᕐᖓᒥᑦ ᓇᓄᖅᓯᐅᕐᓃᑦ, ᑕᓯᐅᔭᕐᒥᙵᕈᖅᓇᖅᑐᑦ.
- **ᑲᑎᕐᒪᔪᑦ** ᐅᕐᑲᖅᑐᑦ ᓇᑦᑎᐊᖅᑖᕐᑲᑦᑕᖅᑐᑦ ᐅᐸᓗᕐᓂᑦ.

ᑲᑎᓪᓚᒃᑖᖅᓗᒋᑦ ᓇᓄᓕᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ ᐅᖃᓪᓗᑲᖅᓴᐅᑎᓃᑦ

- **ᓵᓪ**: ᐱᔪᓐᓇᕐᓂᖃᕐᓂᙵᓄᑦ, ᑕᑯᔪᒪᔭᕗᑦ ᑲᑎᓪᓚᒃᑖᖅᓗᒋᑦ ᓇᓄᓕᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᒥᓱᒃᑯᓇᐊᕐᓗᒋᑦ ᖃᐃᔪᒥᑦ ᓇᓄᖅᓯᐅᕐᓇᐅᓂᐊᖅᑐᒧᑦ.
- **ᐊᓕᓴ** ᐅᖃᐅᓯᖃᖅᑐᖅ ᖃᖓᒃᑰᖓᔪᓂᒃ ᑐᓂᓯᓂᐊᕐᓗᓂ ᐊᒻᒪᓗ ᓄᖅᑲᖓᓂᐅᔪᖕᓇᖅᑐᓂᒃ, ᒪᓐᓃᑦ-ᒥᑦ ᑐᓂᓯᓂᖅ ᐊᔪᕐᓇᕐᓂᐊᖅᑯᖅᑐᖅ.

ᐃᓚᐅᖃᑕᐅᔪᑦ ᖁᔭᓐᓇᒦᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᒪᔫᖃᑕᐅᓚᐅᕋᒪ, ᐊᒻᒪᓗ ᐃᓱᒪᒋᔭᕐᒥᑦ ᑐᓂᔭᕐᒋᓐᓄᑦ.

21:31 ᑲᑎᒪᓂᖅ ᐃᓱᓕᑦᑐᖅ

2021-ᒥᑦ ᐱᙳᙵᖅᐸᓯᙵᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᔪᒥᑦ ᓇᓄᐃᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐅᓂᑉᑳᑉ ᒥᒃᓵᓄᑦ ᐅᖃᖅᑲᑎᖃᕐᓂᕐᒧᑦ ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐊᕆᐊᑦ

ᐃᑲᓂ: ᐊᕆᐊᓂᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᑎᑎᕋᕆᖕᒐᑦ
ᐅᐸᓗᖅ/ᓇᒧᖓᓕᑎᓪᓗᒍ: ᓇᑎᐊᑦ 27, 2024
ᐱᒋᐊᕐᒪᑦ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 19:00
ᐃᓱᓕᑦᑐᖅ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 20:49

ᐃᓚᐅᖃᑕᐅᔪᑦ

ᓂᑯᓪ ᐃᓴᒃᑮᔭᖅ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᑎᑎᕋᖅᑎ)
ᐋᓕᒃᓴ ᐂᓴᓗᒃ (ᐃᒃᓯᕙᐅᑕᖅ) ᐅᖃᖁᑎᒃᑯᑦ
ᑭᐊᕙᓐ ᑲᓗᐊᖅ (ᐃᒃᓯᕙᐅᑕᐅᑉ ᑐᖏᓕᖓ)
ᓘᕋᓐ ᐊᖓᓕᒃ (ᑆᓇᐅᔭᓕᕆᔨ) ᐅᖃᖁᑎᒃᑯᑦ
ᐲᑕ ᓵᒥ (ᑲᑎᒪᔨ)
ᐊᓕᓴ ᐳHᐊᑦ, ᓇᓄᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ
ᐊᕙᓐ ᕆᑦᓱᑦᓴᓐ, ᓇᓄᓂᒃ ᐆᒪᔪᓕᕆᔨ, ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ
ᒥᑦᓴ ᑳᒻᐳᓪ, ᑭᕙᓪᓕᕐᒥᑦ ᓂᕆᔪᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ

ᐱᔾᔪᑎᒋᔭᖓ ᑲᑎᒪᓂᐅᑉ:

ᑐᓂᓯᓗᑎᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓄᑕᐅᓛᓂᒃ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒥᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐱᙳᙵᖅᐸᓯᙵᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᖃᑉᓯᐅᑎᒋᓂᖏᓐᓄᑦ ᐅᖃᐅᓯᐅᓯᒪᓂᙶᑦ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓄᑦ ᐅᓂᑉᑳᒥᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖏᓐᓂᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᑎᓪᓚᒃᑖᖅᓗᒋᑦ ᓇᓄᓕᒃᑕᐅᔪᓐᓇᖅᑐᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᕐᑯᔭᕐᒥᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓱᓇᑐᐃᓐᓇᐃᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐊᐱᖅᑯᑎᒋᔭᐅᔪᑦ ᑕᒪᐃᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᓄᑦ. ᑲᑎᑦᑎᓗᓂ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᑦ ᐱᙳᙵᖅᐸᓯᙵᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᑭᖑᓪᓕᕐᒥᑦ ᐱᓕᕆᐊᖑᓂᐊᖅᑐᑦ ᐊᒻᒪ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ:

ᓇᐃᑦᑐᒥᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (4-6 ᐱᓇᓱᐊᕈᓯᓂᑦ ᑲᒪᓚᐅᖅᑳᖅᑎᓪᓗᒋᑦ):

- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᑲᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᓪᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᐊᐱᖅᑯᑕᐅᔪᑦ
- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᑲᑎᒍᑦ ᓇᑐᓇᐃᖅᑐᒍ ᓇᓚᐅᑦᑎᔪᖅ ᑎᑎᕋᖅᓯᒪᓂᖓᑕ ᑎᕐᒥᓲᒃᑯᑦ ᓈᓴᐃᖃᑕᐅᔪᑦ ᐊᑎᖏᑦ
- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᑲᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᑦᑎᕝᕕᒋᓗᒋᑦ ᖃᑉᓯᓂᒃ ᓇᓄᐃᑦᓴᖅᑖᖅᑎᑕᐅᖃᑦᑕᕐᓂᖏᑦ ᑭᖑᓂᐊᒍᑦ ᐅᑭᐅᖅ 2007-ᒥᑦ ᐅᑯᓄᖓ ᓄᓇᓕᖕᓄᑦ ᐊᕝᕕᐊᑦ, ᑎᑭᕋᖅᔪᐊᖅ, ᑲᖏᖅᖠᓂᖅ, ᐃᒡᓗᓕᒑᕐᔪᒃ, ᐊᒻᒪ ᖃᒪᓂᑦᑐᐊᖅ.

ᐊᑯᓂᐅᓂᓴᒍᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (2-3 ᑕᖅᕿᑦ ᐃᓗᐊᓂᑦ ᑲᒪᓚᐅᖅᑳᖅᑎᓪᓗᒋᑦ):

- ᑐᓂᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐊᐅᓚᓐᓂᖏᓐᓂᒃ ᖁᕝᕙᓯᕈᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ
- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᑲᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᑦᑎᕝᕕᒋᓗᒋᑦ ᑐᖁᕝᕙᔪᓂᒃ ᓇᓄᐃᑦ ᑕᑯᔭᐅᔪᓂᒃ ᓈᓴᐃᑎᓪᓗᒋᑦ ᐅᖃᐅᓯᐅᓚᐅᖅᓂᖏᑦ ᐱᔪᖕᓇᖅᑎᑦᑎᔨᐅᔪᓄᑦ.

ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ:

ᐊᐱᖅᑯᑏᑦ ᑲᑎᖕᒪᔪᓄᑦ ᑎᑭᓯᑎᑕᐅᖅᑳᖅᑎᓪᓗᒋᑦ ᑭᒎᖕᒪᖕᒫᑕ:

ᑲᑎᒪᒋᐊᓚᐅᖅᑐᑦᖃᐅᔨᑎᑕᐅᑉᓗᑎᒃ ᑲᑎᒪᔪᓃᖅᑲᑕᐅᔪᓂᑦ. ᐊᓕᓴ ᐳᕼᐊᑦ ᐊᔾᔨᓂᒃ ᑕᐅᑐᒃᑎᑦᑎᔪᖅ ᐊᓯᐊᓄᐊᖅᐸᓪᓕᐊᑉᓗᑎᒃ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᐃᔨᒃᑯᑦ ᐃᓅᔪᖕᓃᓚᐅᕐᓂᖓᓄᑦ ᐆᒪᔪᓕᕆᔨ ᒫᑲᔅ ᑎᒃ. ᑕᑯᑎᑦᑎᓚᖅᓱᓂ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᐃᓃᑦ ᐃᓚᐅᑉᓗᑎᒃ ᐸᕐᓇᐃᓂᖅᑲᖅᑎᓪᓗᒋᑦ ᑎᑭᔅᓱᒍ ᑲᑎᓱᒃᑖᖅᓱᒋᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᒥᒃᓴᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᓯᒪᓕᐅᕆᓂᖅ ᓈᒻᒪᒋᔭᐅᓚᐅᕐᒪᑦ ᒥᓂᔅᑕᒧᑦ ᐊᒻᒪ ᓇᓄᒍᑏᑦ ᑐᓂᐅᖅᑲᐅᑕᐅᓂᖏᑦ ᓄᓇᓕᖕᓄᑦ. ᐅᖃᖅᓱᓂ ᐱᔾᔪᑎᖕᒐᓂᑦ ᑲᑎᒪᑎᑦᑎᓂᐅᑉ ᑐᓂᓯᓗᑎᒃ ᓇᐃᓈᖅᓯᒪᔪᓂᒃ ᖃᓄᐃᖕᓂᐅᔪᓂᒃ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᐊᖕᒐᓂᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐃᓯᒪᒋᔭᖕᒐᓂᒃ ᑐᓴᕐᓂᐊᕐᓗᑎᒃ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᑦ ᐊᔾᔨᑦ

- ᐅᖃᐅᓯᒃᓴᓂᒃ ᑐᓂᓯᔪᖃᓚᐅᖏᑦᑐᖅ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᔪᑦ/ᐱᓕᕆᓃᑦ

- **ᑭᐊᕙᓐ**: ᑕᓯᐅᔭᕐᔪᐊᑉ ᑕᕆᐅᖓ ᐱᓕᕆᐊᕆᒍᖕᓂ, ᓇᒧᑦ ᓄᓇᖅᐸᓯᒃᑐᒧᑦ ᑎᑭᐅᑎᖅᑕᖅᐱᑦ?
- **ᐊᓕᓴ**: ᓯᒡᔭᑎᒍᑦ ᖃᖓᑕᖅᑕᖅᓯᓂᐊᖅᑐᒍᑦ ᐊᒻᒪ ᕿᓂᖅᑐᑦ ᑎᕐᒥᓲᕐᒥᖅᓱᑎᒃ ᑕᑯᔪᖕᓇᖅᑐᑦ ᓯᒡᔭᓂᑦ ᑕᕆᐅᕐᒧᑦ, ᑭᓯᐊᓂ, ᐅᖓᓯᒃᑲᓐᓂᖅᓂᖅᓴᐅᔪᒧᑦ ᑕᕆᐅᕐᒥᑦ ᑕᑯᓂᖅ ᐊᔪᕐᓇᖅᑕᖅᑐᖅ. ᐱᔾᔪᑕᐅᓗᐊᖅᑐᖅ ᐃᒫᓂᑦ ᓈᓴᐃᖅᑕᕐᓂᖅᓴᓂᒃᑎᖕᓄᑦ ᐊᑦᑕᓴᓇᖅᑐᒦᔅᑯᖕᒋᓴᖅᒐᓕᑦ ᐃᑭᒪᖅᑕᐅᔪᑦ ᑎᕐᒥᓲᕐᒥ. ᐱᔪᒥᓇᖅᑑᔪᖅ, ᐆᒃᑐᕋᖅᑕᐅᓗᑎᒃ ᐃᓄᒃᑕᐅᓯᒪᖕᒋᑦᑐᑦ ᖃᖓᑕᓲ. ᐊᑐᖅᑕᐅᖅᑲᑦᑕᖅᑐᑦ ᓯᕗᓪᓗᒃᓯᒪᒃᑲᓐᓂᓕᖅᐸᑕ, ᐊᑐᕈᓐᓇᕐᓂᐊᖅᑕᕗᑦ ᐃᓄᒃᑕᐅᓯᒪᖕᒋᑦᑐᑦ ᖃᖓᑕᓲ ᓈᓴᐃᓂᐊᕐᓗᑕ ᑕᕆᐅᕐᒥᑦ, ᐊᒻᒪ ᒥᑦᔅ ᐱᓕᕆᐊᕐᒥᑦ ᓴᖅᑭᑦᑎᓂᐊᖅᑐᖅ ᐃᓄᒃᑕᐅᓯᒪᖕᒋᑦᑐᓂᒃ ᖃᖓᑕᓲᓂᒃ.
- **ᒥᑦᔅ**: ᓄᓇᕗᒻᒥᑦ, ᖃᖓᑕᖃᑦᑕᖅᑐᑦ ᐃᒫᓄᑦ ᕿᑭᖅᑕᓂᑦ ᓈᓴᐃᓂᐊᕐᓗᑕ ᖃᓂᑦᑐᒦᑦᑐᓂᑦ, ᑲᔪᓯᓇᔭᖅᑐᒍᑦ. ᐃᒪᑲᓪᓚᖕᒥᑦ ᖃᐅᔨᓴᐃᖃᑦᑕᖅᑐᑦ ᐊᒻᒪ ᒥᑭᓂᖅᓴᖅ ᑎᕐᒥᓲᖑᓚᐅᖅᑐᖅ, ᑐᐃᓐ ᐋᑐᒥᑦ ᐊᑐᖅᓗᑕ, ᐃᒪᒃᑯᑦ ᖃᖓᑕᔪᖕᓇᖅᑐᒍᑦ, ᐅᐊᖕᓇᑦᑎᐊᖕᒐᓂᑦ ᑎᑭᓯᔪᐊᑉ ᕿᑭᖅᑕᖅᑲᖅᑐᓂᑦ. 2016-ᒥᑦ, ᒪᕿᖕᓂᒃ ᐃᑯᒪᑖᖕᓂᑦ ᖁᓕᒥᒎᓕᖅᑲᓚᐅᖅᑐᑦ ᑕᐃᒪᓐᓇᒧᑦ ᐅᖓᓯᖕᓂᖅᓴᒨᕐᒐᐅᔪᖕᓇᓚᐅᖅᑐᒍᑦ ᐃᒪᕐᒥᑦ 2016-ᒥᑦᑕᐅᖅ.
- ᓄᑖᓂᒃ ᐃᓄᒃᑕᐅᓯᒪᖕᒋᑦᑐᓂᒃ ᖃᖓᑕᓲᖅᑕᖅᑲᓕᖅᑐᒍᑦ. ᐱᑕᖅᑲᕈᓐᓇᖅᑐᑦ ᑎᑭᓪᓗᒍ, ᐱᑐᐊᖅᑐᒥᑦ ᐊᐅᔭᒃᑯᑦ 20-ᓂᒃ ᐃᑲᖅᑯᓄᑦ, ᖃᖓᑕᖅᑲᑦᑕᕈᓐᓇᖅᑐᑦ ᐃᖕᒥᖕᓂᑦ ᐊᒻᒪ ᐃᑲᖅᑯᐃᑦ ᐊᕙᑎᓄᑦ ᑎᑭᓪᓗᒍ. ᐆᒃᑐᖅᑭᒪᔭᕗᑦ ᐊᐅᔭᐅᔪᖅ.
- **ᑭᐊᕙᓐ**: ᐅᒥᐊᒃᑰᖅᑎᕐᓴᖅᓕ ᓯᓴᒃᑯᑦ ᑰᒡᔪᐊᕐᒧᑦ, ᑕᑯᓂᐊᖅᑐᒍᑦ ᐱᖕᓯᓂᑦ ᐅᕙᓐᓂᖕᓂᑦ ᓯᑕᒪᓂᑦ ᓇᓄᕐᓂᑦ ᐊᑕᐅᓯᕐᒥᑦ ᐅᒥᐊᒃᑰᖅᑎᒋᓪᓗᑕ. ᐊᖏᓂᖅᓴᐅᓕᔪᑦ. ᒥᑭᔪᒥᒃ ᖃᑯᖅᑐᒥᒃ ᑕᑯᔪᖕᓇᑐᐃᓐᓇᖅᑐᓇᖅᑐᑎᑦ ᖁᓕᒥᒎᓕᒃᑯᑦ. ᐃᓄᒃᑕᐅᓯᒪᖕᒋᑦᑐᑦ ᖃᖓᑕᓲ ᐊᔪᖕᒋᑉᐸᑕ, ᑕᑯᔪᖕᓇᕐᓂᐊᖅᑐᑦ ᐊᑐᖅᑲᑦᑕᖅᑕᑉᑎᖕᓂ ᐅᖓᓯᒃᑐᒧᑦ ᐊᐅᓚᖅᓯᒪᓴᖏᖅᓯᓪᓗᑕ.
- **ᒥᑦᔅ**: ᐊᔾᔨᑦ ᑕᑯᒃᓴᐅᑎᐊᖅᑑᔪᑦ, ᖃᐅᔨᔪᖕᓇᖅᓯᓂᐊᖅᑐᑎᑦ ᓇᓄᕐᓂᑦ ᓇᑐᓇᖕᒋᑦᑐᑦ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᖃᓄᐃᖕᓂᐅᔪᑦ

- **ᐋᓚᒃᔅ**: ᒪᕿᖕᓂᒃ ᐊᐱᖅᑯᑎᒃᓴᖅᑲᖅᑐᖓ. ᖃᐃᔪᐃᕙᐅ ᓇᓄᒍᑎᒃᓴᐃᑦ ᐅᑭᐅᑕᒫᑦ ᑐᓂᔭᐅᔪᑦ ᑭᖑᓂᐊᒍᑦ ᐅᑭᐅᖅ 2007-ᒥᑦ ᓄᓇᓕᖕᓄᑦ ᐊᕝᕕᐊᑦ, ᑎᑭᕋᖅᔪᐊᖅ, ᑲᖏᖅᖠᓂᖅ, ᐃᒡᓗᓕᒑᕐᔪᒃ, ᐊᒻᒪ ᖃᒪᓂᑦᑐᐊᕐᒧᑦ?
- ᑕᐃᒪᙵᓂᑦ 1950-ᖏᓐᓂᑦ, 1960-ᖏᓐᓂᑦ, 1970-ᖏᓐᓂᑦ, ᓄᓇᓕᒃᑎᒍᑦ ᑭᕙᓪᓕᕐᒥᑦ ᒪᓕᖅᑕᖅᓯᒪᔪᒍᑦ ᓇᓄᒃᑕᐅᖅᑯᔭᐅᔪᓂᒃ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓄᑦ 3:1-ᖑᑉᓗᓂ, ᑐᑭᖅᑲᖅᑐᖅ ᐅᑭᐅᖑᓚᐅᖅᑐᑐᒃᑖᑦ ᓇᓄᖅᑲᖅᑕᖅᑕᕈᑦ ᐊᖑᓴᓪᓗᐃᓐᓇᐅᔪᑦ. ᒪᓕᖅᑲᑦᑕᖅᑕᕈᑦ ᒪᓕᒐᑦᓴᑦ ᒪᓕᖅᑯᔭᐅᔪᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓄᑦ 35 ᐅᑭᐅᓄᑦ. ᐅᑉᓗᐃᑦ

ᐊᓂᒍᖅᑐᓂᑦ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐅᖃᓚᐅᖅᑐᑦ ᐊᒥᓲᔪᖕᓃᖅᑐᑦ ᐊᒻᒪᓗ ᐃᓱᒪᑉᓗᑎᒃ ᐊᖑᓵᓪᓗᐃᑦ ᓇᓄᒃᑕᕆᔭᐅᓗᐊᖃᑦᑕᕐᓂᖅᑭᖕᓄᑦ. ᑕᐃᒪᖔᓂᑦ 2015-ᒥᑦ, ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ 1:1-ᒧᑦ ᐊᓯᐊᙵᖅᑕᐅᓚᐅᖅᑐᑦ. ᑕᐃᒪᓐᓇ ᐅᖃᖅᑎᓪᓗᑕ, ᐅᕙᒍᑦ, ᓄᓇᑖᖃᑕᐅᓯᒪᔪᒍᑦ, ᐊᒥᓲᔪᖕᓃᖅᑎᙱᑕᕗᑦ, ᑭᓯᐊᓂ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑕᐃᑉᓱᒪᓂ 30 ᐅᑭᐅᑦ ᐅᖓᑖᓄᑦ, ᐅᖃᖃᑦᑕᖅᑐᑦ ᓇᓄᐃᑦ ᐅᒥᓲᔪᖕᓃᖅᑐᑦ.

- ᐅᖃᐅᓯᑐᐃᓐᓇᓗᐊᖑᔪᖅ, ᑐᓂᔭᐅᒫᖓᑕ 38-ᓂᑦ ᑲᑎᓗᒃᑖᖅᐳᕆᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ ᓄᓇᓖᑦ ᐊᕐᕕᐊᑦ, ᑎᑭᕋᔪᐊᖅ, ᐊᒻᒪᓗ ᑲᖕᒥᖅᓱᓂᖅ, ᑕᐃᒪᙵᓂᑦ 2007-ᒥᑦ, ᓲᕐᓗ ᐅᓇᑕᕈᑎᒋᔭᕗᑦ 38 ᓇᓄᒍᑏᑦ, ᑕᐃᒪᓐᓇ ᐃᓄᐃᑦ ᐱᖅᑯᓯᕆᙱᑕᖓᑦ ᐅᓇᑕᖃᑦᑕᐅᑎᓗᑎᒃ. ᐊᕐᕕᐊᑦ ᓄᓇᓕᕐᔪᐊᖑᑉᓗᓂ, ᐊᒥᓱᒻᒪᕆᐊᓗᖕᓂᑦ ᓇᓄᕐᓄᑦ ᑎᑭᑕᐅᖃᑦᑕᖅᑐᒍᑦ ᐅᑭᐅᑕᒫᒃᑯᑦ ᑕᐃᒪᙵᓂᑦ 2007-ᒥᑦ. ᐊᔾᔨᖓᓂᒃ ᑲᖕᒥᖅᓴᓂᐅᑉ ᐊᒻᒪᓗ ᑎᑭᕋᔪᐊᑉ ᐱᖃᑦᑕᖅᑐᒍᑦ. ᐃᓱᒪᔪᖓ ᐊᕐᕕᐊᑦ ᐃᒥᓵᓂᖅᓴᓂᒃ ᑲᑎᓗᒃᑖᖅᐳᕆᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ ᑐᓂᔭᐅᔭᕆᐊᖃᖅᑐᖅ.
- **ᐊᓕᓴ:** ᐱᔾᔪᑎᒋᑉᓗᒍ ᓯᕗᓪᓕᖅ ᐊᐱᖅᑯᑎᒋᖅᖃᐅᔭᐃᑦ, ᖃᐅᔨᒋᐊᕐᓂᐊᖅᐸᒃᑲ ᑎᑎᖅᖃᐅᑎᕗᑦ ᐊᒻᒪᓗ ᐱᔪᖕᓇᖅᑕᒃᑲ ᓈᓴᐅᑏᑦ ᐊᒻᒪᓗ ᖃᓇᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᑲᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᓪᓗᕆᑦ ᓂᑳᓪ-ᒧᑦ ᐃᓕᑉᓯᖕᓄᐊᕐᓂᐊᕐᓗᓂᐅᒃ.
- ᐱᑦᑕᐅᖕᒋᑦᑐᒥᒃ ᖃᓄᐃᖓᓕᖅᑎᑦᑎᓂᖃᕈᖕᓇᕐᒥᔪᖅ ᐊᒥᓱᓗᐊᑦ ᐊᖑᓵᓪᓗᐃᑦ ᓇᓄᒃᑕᐅᖃᑦᑕᖅᐸᑕ, ᒪᑦᓇ ᑕᒪᑐᒥᙵᑦ ᑐᓂᓯᒐᕙᑦ, ᐋᓕᒃᓴ. ᖃᐅᔨᑎᑦᑎᓂᐊᖅᐳᖓ ᐊᑕᐅᓯᕐᒥᑦ ᐊᒻᒪᓗ ᐱᖃᑖᓂᒃ ᐊᑕᐅᓯᕐᒥᒃ ᓇᓄᒍᖕᓇᕋᓱᐊᑕᐅᓯᕐᒥᑦ ᐊᖑᓵᓪᓗᕐᒥᑦ ᐊᕐᓇᓪᓗᓗᒃᑖᓄᑦ. ᑭᓯᐊᓂ ᐊᒥᓲᓂᖅᓴᓂᒃ ᐊᖑᓵᓪᓗᕐᓂᑦ ᓇᓄᒍᒪᒍᑉᓯ ᐊᕐᓇᓪᓗᓂᐅᖔᓂᑦ, ᐊᔪᙱᑎᑕᐅᖕᒥᔪᖅ ᐊᑕᐅᓯᕐᒥᑦ ᐊᑕᐅᓯᕐᒧᑦ ᐋᖅᑭᐅᒪᔪᒥᑦ. ᓇᓄᒍᒪᒍᑉᓯ 12-ᓂᑦ, 8 ᐊᖑᓵᓪᓗᐃᑦ ᐊᒥᐊᒃᑯᖕᒋᑦ ᐊᕐᓇᓪᔫᓗᕆᑦ, ᑕᐃᒪᓐᓇᐃᑦᑐᖕᓇᖅᑐᓯ ᐅᕝᕙᓘᓐᓃᑦ ᐊᕐᕕᓂᓖᑦ ᐊᖑᓵᓪᓗᕐᓂᑦ, ᐊᕐᕕᓂᓖᑦ ᐊᕐᓇᓪᓗᓂᒃ. ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐃᓱᒪᓕᐅᕈᖕᓇᖅᑐᑦ.

- **ᑭᐊᕕᓐ:** ᑐᓴᐸᓗᒍ ᐊᒻᒪᓗ ᑕᐅᑐᒃᐳᒍ ᐊᔾᔾᓃᑦ ᑕᑯᒃᓴᐅᑎᑕᓯ, ᓲᕐᓗ ᖃᖓᑕᖅᑲᑦᑕᖅᑐᓯ ᐊᒻᒪᓗ ᐃᒃᓯᕙᐸᓗᓯ ᖃᓄᑕᐅᔭᕐᒥᑦ, ᓈᓴᐅᓯᓇᓕᖅᐳᓯ ᐃᓱᒪᓕᐅᕐᓂᐊᕐᓗᓯ ᖃᐸᓯᐅᓂᕐᒪᖕᓖᑦ ᑲᑎᓗᒃᑖᖅᐳᕆᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐱᖃᑦᑕᖅᑕᐸᑎᖕᓂᑦ, ᓱᓕᕙ?
- ᖃᓄᖅ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐃᑲᔪᖃᑦᑕᖅᐸᑦ ᓈᓴᐅᑏᑦ ᐱᓂᐊᕐᓗᒋᑦ?
- 8-ᖑᔪᓂᒃ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᒪᓕᒐᖅᑐᑦ. ᐃᓚᖓᑦ ᐃᒃᐱᒍᓱᙱᕐᓂᖅ ᐱᖅᑯᓯᖓᓐᓂᑦ ᐊᒻᒪᓗ ᐊᕙᑎᖓᓃᑦᑐᓂᑦ. ᐅᖃᐅᓯᐅᖃᑦᑕᖅᑐᖅ ᐃᓐᓇᓇᔭᑎᖓᓂᑦ, ᓇᓕᐊᑐᐃᓐᓇᐅᕙᓗᐊᖅᐸᑦ ᓄᓇᓕᒃ, ᐃᓂᐅᔪᖅ ᐅᕝᕙᓘᓐᓃᑦ ᐅᐊᖓᖅᐸᓯᒃᑐᒥᑦ ᓄᓇᒋᔭᐅᔪᒨᕙᐅᒍᕕᑦ (ᐊᓛᒃ, ᐊᑯᑭᑦᑐᖅ), ᐅᖃᐃᓐᓇᖅᑲᑦᑕᖅᑐᑦ, ᓂᕆᑎᓂᒃ ᐱᑕᖅᑲᑦᑕᖅᑐᑦ ᐊᒥᓱᓗᐊᖑᑎᙱᓪᓗᒋᑦ. ᓲᕐᓗ ᐃᓱᒫᓗᑎᑕᖅᑲᙱᑦᑐᖅ ᓇᓄᐃᑦ ᐊᒥᓱᒻᒪᕆᐅᑎᓐᓂᐊᕐᓗᒋᑦ. ᐅᖃᙱᑦᑐᖓ ᑐᕐᑯᓗᒃᑖᕐᓗᒋᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᒥᓲᔪᖕᓃᖅᑎᑦᑎᐊᕐᓗᒋᑦ, ᑭᓯᐊᓂ ᐃᒃᐱᒍᓱᙱᕆᓯᐅᒃ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐊᒻᒪᓗ ᑕᑯᒃᓴᐅᑎᓪᓗᒋᑦ ᖃᓄᖅ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐃᓚᐅᑎᑕᐅᖃᑦᑕᕐᒪᖔᑕ ᖃᐅᔨᓴᐃᓂᐊᕐᓯᖓᓂᑦ.
- **ᑭᐊᕕᓐ:** ᖃᐅᔨᓴᐃᓂᕗᑦ ᐱᖓᖕᓇᖅᐸᓯᒃᑐᒃᑯᑦ ᖃᐸᓗᓈᓂᑦ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᐅᔪᖅ, ᑭᓯᐊᓂ ᐃᓚᐅᑎᖃᑦᑕᖅᑕᕗᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐃᑲᔪᕐᓂᐊᕐᓗᑎᒃ ᐃᓂᐅᔪᓂᒃ ᑲᒪᒋᔭᐸᑎᖓᓂᑦ ᐊᒻᒪᓗ ᖃᐅᔨᒪᓂᕆᔭᐅᔪᑦ ᓇᓄᐃᑦ ᓇᓃᓐᓂᐊᕐᒪᖔᑕ. ᓄᓇᓕᖕᒥᑦ ᐃᓚᐅᔪᕐᖅᑐᑦ ᓈᓴᐃᖅᑕᐅᔪᓂᑦᑕᐅᖅ. ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᒃ ᖃᐅᔨᓴᐃᓇᓱᒃᑲᖅᓯᒪᔪᒍᑦ ᓇᓄᐃᑦ ᒥᒃᓵᓄᑦ ᐱᓕᕆᐊᑉᑎᖓᓂᑦ, ᐊᔪᙱᒃᑳᖓᑉᐸ. ᐊᓯᖓᑦ ᐊᐅᓚᑦᑎᔨᐅᔪᑦ ᑲᑐᔾᔨᖃᑎᒌᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᒃ ᖃᐅᔨᓴᐃᓂᐊᖅᑐᑦ ᓄᓇᓕᖓᓂᑦ ᐊᒻᒪᓗ ᑲᑎᑎᓗᑎᒃ ᖃᐅᔨᒪᓂᕆᔭᐅᔪᓂᑦ. ᑕᐃᒪᓐᓇᐃᓚᐅᖅᑐᒍᑦ Davis Strait-ᒥᑦ, ᑐᑭᓯᐊᓕᖅᑎᑦᑎᕚᓪᓕᖅᐳᓂ ᐅᕙᑎᖓᓂᑦ ᓇᓄᐃᑦ ᐊᒥᓲᑎᕆᓂᖓᓐᓂᑦ ᑕᒪᐃᓐᓂᑦ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖅᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᙱᑦ ᐊᑐᓚᐅᕋᑉᑎᒍᑦ. ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖅ ᑭᓪᓕᖅᑲᖅᑐᖅ ᓇᐃᑑᔪᒧᑦ ᐱᕕᖃᕐᓂᕐᒥᒃ ᓄᓇᒧᐊᖅᑲᑦᑕᖅᐳᑎᒃ ᐅᑭᐅᑕᒫᒃᑯᑦ ᐊᒻᒪᓗ ᐱᒋᐊᓕᓵᓚᐅᖅᑐᑦ 1980-ᖑᓐᓂᑦ, ᒪᖃᐃᑏᓪᓕ ᓄᓇᒨᕙᐅᓯᒪᖃᑦᑕᖅᑐᑦ ᐅᑭᐅᑉ ᐃᓗᐊᓂᓗᒃᑖᕐᓚᒃ, ᑕᑯᖃᑦᑕᖅᐳᑎᒃ. ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐊᕙᙵᑲᓛᓗᖕᒥᑦ ᓯᕗᓕᓇᔭᐅᔪᓂᑦ ᐱᓯᒪᔪᑦ ᐅᑭᐅᑦ ᐊᒥᓱᒃᑳᓗᖕᓄᑦ. ᐱᖃᕐᓗᓂ ᑕᒪᐃᓐᓂᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒥ ᐃᓗᐃᑦᑑᖓᓂᖅᓴᐅᔪᒥᒃ ᑐᑭᐊᑎᑦᑎᓂᖅᑲᖅᑐᖅ ᐅᕙᑎᖓᓄᑦ.
- ᓈᓴᐅᑏᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᓂᒃ ᑐᓂᓯᓂᐊᖅᑐᑦ ᐊᒻᒪᓗ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ ᐊᑐᖅᐳᕆᑦ, ᐃᓱᒪᒋᓂᐊᖅᑕᕗᑦ ᑕᒪᕐᒥᓗᒃᑖᖅ ᐱᔾᔪᑕᐅᔪᑦ ᐅᖃᐅᓯᕆᐅᖅᑲᒃᑲᑦ ᓲᕐᓗ ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᖑᕆᓪᓗᑎᒃ, ᐊᓯᐅᔨᑦᑕᐃᓕᓂᖅ, ᐊᒻᒪᓗ ᑕᑯᓇᓱᖅᑲᑦᑕᖅᑕᕗᑦ ᖃᓄᐃᓕᐅᖅᐸᓪᓕᐊᓂᐅᔪᑦ, ᖃᐱᐅᓂᖕᒋᐅᓗᐊᖑᙵᕆᑦᑐᑦ ᓇᓄᐃᑦ. ᑕᑯᐸᑎᒍᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑲᑎᕐᓗᕆᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ ᐊᑯᓐᓂᖓᓐᓂ 2016 ᐊᒻᒪᓗ 2021, ᓲᕐᓗ ᖃᐱᐅᓂᖏᑦ ᓱᕐᑯᓯᙱᑦᑐᑦ, ᐱᒋᐊᖅᑎᑦᑎᓇᔭᖅᑐᖅ ᐱᕐᑯᔨᓂᓇᔭᑉᑎᖓᓂᑦ ᑲᑎᓗᒃᑖᖅᐳᕆᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᓱᕐᑯᓯᓇᐊᖅᑲᙱᕐᓗᑎᒃ.
- ᐱᓐᓂᑰᔪᑦ 38-ᓂᒃ ᑲᑎᓗᒃᑖᖅᐳᕆᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐱᖓᖕᓇᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓈᓴᐅᑎᒥᙱᓚᐅᖅᑐᖅ 4.5%-ᒥᑦ ᓈᓴᐅᑎᖅᑲᖅᑐᖅ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᒃᑯᑦ ᓈᓴᐅᓯᓇᔾᔪᑕᐅᓚᐅᖅᑐᒥᑦ, ᑕᒫᓂᓕᖅᖄᖅᑎᓐᓇᖓ, ᐊᔪᙱᑦᑐᒃᑯᑦ ᒪᖃᐃᑕᐅᔪᖕᓇᖅᑐᓄᑦ ᐳᓴᓐᑎᐅᔪᖅ ᖃᐱᐅᓂᖕᒋᓐᓄᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ. ᑕᐃᒪᓐᓇᒧᓪᓗ 38 ᐃᒪᓐᓇᐃᑦᑐᖅ 4.5%-ᖑᔪᖅ 842-ᓂᑦ.
- ᑐᕋᒐᓇᔭᐅᔪᓂᑦ ᐃᓚᒋᔭᐅᔪᖅ ᐅᖃᖅᑲᑎᖅᑲᕐᓂᕐᒥᑦ ᐅᖃᐅᓯᕆᓂᐊᕐᓗᒍ ᐆᒃᑑᑕᐅᔪᖅ ᑲᑎᓗᒃᑖᖅᐳᕆᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ 38-ᓂᒃ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ ᐊᒻᒪᓗ ᐱᓂᐊᕐᓗᒋᑦ ᑲᑎᒪᔾᔪᑏᑦ, ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐃᓱᒪᒋᔭᖏᓐᓂᑦ ᓈᓴᐅᑎᐅᔫᑉ ᒥᒃᓵᓄᑦ. ᐱᒻᒪᕆᐅᖕᒥᔪᖅ ᐅᕙᑎᖕᓄᑦ ᑐᓂᓯᔾᔪᑎᓗᑕ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ, ᖁᔭᓐᓇᒦᑦ ᑐᓂᓯᔪᖕᓇᖅᖃᐅᒐᑉᓯ ᐃᓚᖓᓐᓂᑦ ᒪᓕᒐᓂᑦ ᐊᒥᓱᓗᐊᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ.

- **ᑭᐊᘄᓐ** ᓇᓗᓇᐃᖅᑐᖅ ᐃᒪᓐᓇᐃᖅᑯᔨᓗᓂᓗᐊᖑᙱᑦᑐᖅ ᐊᒥᓲᔫᖑᓃᕐᓗᒋᑦ, ᑐᓂᓯᖅᑲᐅᔪᖅ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᕐᒋᓐᓂᑦ.

- **ᓘᐊᓐᓇ**: ᑭᓖᑕᓐ NIWS-ᑯᓐᓂᑦ ᐅᖃᐅᑎᓚᐅᖅᑖᑎᒍᑦ ᐊᐱᖅᓯᐃᓂᖅᑕᖃᓚᐅᖅᑐᖅ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᕐᒋᓐᓄᑦ ᐊᕿᐊᓂᑦ ᐊᐱᖅᓱᖅᓱᑎᒃ ᓇᓄᐃᑦ ᐊᒥᓲᑎᒋᓂᕐᒋᓐᓂᑦ. ᐃᓕᕝᓯᖕᓄᐊᖅᑕᐅᓚᐅᖅ? 2015-ᒥᐅᓚᐅᖅᑐᖅ, 2014-ᒥᑦ, ᐃᖅᑲᐃᒪᙱᑦᑕᕋ ᐅᑭᐅᖅ.
- ᐃᖅᑲᐅᒪᔪᕐᓕ ᐋᓖᒃᓴ ᐅᖃᖅᑐᖅ 2021 ᐅᑭᐅᖅ ᐅᕙᑎᖕᓄᑦ ᐱᑦᑕᐅᕐᒋᑦᑑᓚᐅᕐᓂᕐᓗᓄᑦ, ᑲᓇᕐᓇᖅᐸᓯᒃᑐᓂᑦ ᐊᓄᓈᖃᖃᑦᑕᖅᓗᓂ ᐊᒻᒪᓗ ᓯᓚᕐᑰᑕᐅᖃᑦᑕᖅᓗᓂ ᑕᒪᒃᑯᐊᕋᓗᐃᑦ – ᓈᓴᐃᑎᓪᓗᒋᑦ ᓇᓄᕐᓂ ᓄᓇᑉᑕ ᖃᓂᑖᓂᑦ. ᑕᐅᑐᒃᓗᒍ ᐱᕐᒪᕐᓇᖅᐸᓯᕐᒪᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒥᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒥᑦ, ᓲᕐᓗ ᑭᐳᒃᑑᒃ, ᐊᒻᒪᓗ ᑲᑎᓯᒪᔪᑦ ᓈᓴᐅᑏᑦ, ᐊᔾᔨᒌᙱᓐᓂᖅᑕᖅᑐᖅ ᐊᑕᐅᓯᕐᒥᑦ ᓇᓄᕐᒥᑦ, ᑖᒻᓇ ᒥᒃᓵᒋᐊᖅᓯᓂᐅᓂᐊᖅ?
- **ᐊᓕᓴ**: ᑐᓴᓚᐅᖅᓯᒪᔪᕐᓕ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᕐᒋᓐᓂᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᒃ ᐸᕐᓇᐅᑕᐅᓚᐅᖅᑐᖅ, ᑭᓯᐊᓂ ᐅᕙᑎᖕᓄᐊᖅᓯᔪᖃᓚᐅᕐᒋᑦᑐᖅ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᑲᓂᒃ. ᐱᒻᒪᕆᐅᓇᔭᖅᑐᖅ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᑐᓂᔭᐅᓗᑎᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ.
- ᐊᓕᐊᓇᐃᑦ ᑕᒪᓐᓇ ᐅᖃᐅᓯᕆᓚᖓᓂ, ᑐᓴᖅᓯᒪᒐᑉᑕ ᐊᓯᖏᓐᓂᑦ ᓄᓇᓕᖕᓂᑦ 2021 ᐊᔾᔨᐅᖏᑦᑑᓚᐅᕐᓂᖓ ᓯᓚᐅᑉ ᐊᒻᒪᓗ ᐊᕙᓐ ᐅᖃᐅᓯᖄᔪᖕᓂᐊᖅᑐᖅ ᖃᓄᖅ ᓯᑯ ᐊᔾᔨᐅᖏᑦᑑᓚᐅᕐᒪᖔᑦ ᑕᐃᑲᓂ ᐅᑭᐅᕐᒥᑦ.
- ᐱᔾᔪᑎᒋᐸᓗᒍ ᑲᑎᓯᒪᐸᓗᑎᒃ ᐱᕐᒪᕐᓇᖅᐸᓯᕐᒪᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒥᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒥᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ 2016-ᒥᑦ 2021-ᒧᑦ, ᐃᓱᒪᒋᓇᔭᙱᑕᕗᑦ ᐊᔾᔨᒌᙱᒋᓐᓂᖃᖅᑐᖅ ᐊᑕᐅᓯᕐᒥᑦ ᒥᒃᓵᒋᐊᕐᓂᕐᒋᓐᓄᑦ ᐊᒥᓲᑎᒋᓂᕐᒋᑦ, ᖃᐸᓯᐅᑎᒋᓂᕐᒋᑦ ᐃᓱᒪᒋᓇᔭᖅᑕᕗᑦ ᓱᖁᓯᐊᕐᒋᑦᑐᑐᑦ.

- **ᐲᑕ**: ᐋᓐᓂᐊᖅᑐᓂᒃ ᓇᓄᕐᓂᑦ ᑕᖁᖃᑦᑕᖅᐱᓯ ᓈᓴᐃᑎᓪᓗᓯ? ᐅᖃᐅᓯᕆᓇᔭᖅᐱᓯᐅᒃ ᓄᓇᕗᑦ ᑐᙱᕕᒃᑯᓐᓄᑦ? ᐊᒥᓲᔫᖅᑎᑦᑎᖃᑦᑕᖅᐸᑦ ᓇᓐᓄᒃᑕᐅᔪᕐᓇᖅᑐᓂᒃ ᓄᓇᓕᕐᓄᑦ?
- **ᐊᓕᓴ**: ᓈᓴᐃᓂᖅ ᐅᓂᑉᑳᒦᑦᑐᖅ, ᓯᑕᒪᓂᒃ ᑐᖁᕐᒪᔪᓂᒃ ᓇᓄᕐᓂᑦ ᑕᑯᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ ᐆᒃᑐᕋᕈᑕᐅᓂᐊᖅᑐᓂᒃ ᐱᖒᐅᓚᐅᕐᒋᑦᑐᑦ, ᖃᐅᔨᒪᙱᑦᑕᕗᑦ ᓱᓇᒥᒃ ᐱᔾᔪᑎᖃᖅᓗᑎᒃ ᑐᖁᓚᐅᕐᒪᖔᑕ ᓇᓄᐃᑦ. ᐋᓐᓂᐊᕐᓂᕐᒨᔪᕐᓇᖅᑐᖅ ᐅᕙᓘᓐᓃᑦ ᐱᕐᓕᕐᒪᑦ ᐅᕙᓘᓐᓃᑦ ᐅᑭᐅᖅᑯᖅᑐᓕᕐᓂᕐᓗᓄᑦ. ᐅᖃᐅᓯᕆᔭᐅᖅᑯᓚᐅᖅᑐᖅ ᐃᓚᖓᓐᓂᑦ ᓇᓄᕐᓂᑦ 21-ᓂᒃ ᐅᑭᐅᖃᖅᓗᓂ ᐊᖑᓴᓪᓗᖅ. ᐃᓐᓇᐅᓂᖅᓴᐅᔪᖅ ᓇᓄᕐᓄᑦ, ᐊᔾᔨᐅᕐᒋᑦᑐᐊᓘᓗᐊᕐᒋᑦᑐᖅ. ᖃᓄᐃᓕᐅᖃᑦᑕᓚᐅᕐᒪᖔᑕ ᖃᐅᔨᒪᓗᐊᙱᑦᑐᕐᓕ, ᐅᖃᐅᓯᐅᓚᐅᕐᒪᖔᑦ ᑭᓇᑐᐃᓐᓇᕐᒧᑦ, ᐊᐱᕆᔪᕐᓇᖅᑐᕐᓕ.
- ᐃᖕᒥᓂᒃ ᑐᖁᔪᒥᒃ ᓇᓄᕐᒥᑦ ᓇᓂᔭᐅᔪᖃᙵᕐᓕᑦ, ᓇᓐᓄᒐᒃᓴᐅᔪᓂᑦ ᐲᖅᑕᐅᖃᑦᑕᙱᑦᑐᖅ.
- **ᒥᑭ**: ᓄᓇᕗᒃᑰᓚᐅᖅᑐᒍᑦ ᖃᕐᒪᑦᑖᓗᑕ ᐊᒻᒪᓗ ᓇᓄᐃᑦ ᑕᑯᔭᓗᒃᑖᕗᑦ ᖃᓄᐃᑑᔭᓚᐅᕐᒋᑦᑐᑦ, ᓯᑏᕙᓐ ᒫᓂᑑᐸᒃᑯᑦ ᖃᕐᒪᑦᑕᓚᐅᖅᑐᖅ ᑕᐅᕙᓂ ᑐᖁᕐᒪᔪᑦ ᓇᓄᐃᑦ ᑕᑯᔭᐅᐸᓗᑎᒃ.

ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ (ECCC) ᖃᐅᔨᓴᐃᓂᕐᓗᑦ

- ᐊᓕᓴ ᐊᐅᓚᔪᓂᒃ ᑕᑯᑎᑦᑎᔪᒪᓚᐅᖅᑐᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓂᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒥᙳᖅᑐᑦ ᖁᕐᓗᓯᕈᑏᑦ ᐊᐅᓚᔪᒧᐊᖅᑕᐅᓚᐅᖅᑐᑦ.
- **ᐋᓕᒃᓴ**: ᓄᓇᕗᒻᒥ ᓚᕙᒪᒃᑯᑦ ᖃᕐᓗᑖᒃᑯᑦ ᓈᓴᐃᓚᐅᖅᑐᑦ 2006-ᒥᑦ ᐊᒻᒪᓗ 2007-ᒥᑦ, ᑕᐃᑉᓱᒪᓂ ᑲᑎᒪᔨᐅᖃᑕᐅᓚᐅᖅᑐᕐᓗ. ᑕᑯᓚᐅᖅᑕᕗᑦ ᐅᓂᑳᓕᐊᕐᒋᑦ ᓄᓇᕗᒻᒥ ᓚᕙᒪᒃᑯᑦ. ᐆᒪᔪᓕᕆᔨ ᑕᐃᑉᓱᒪᓂ 14 ᐅᐸᓗᓄᑦ ᖃᕐᓗᑖᒃᑯᑦ ᓈᓴᐃᔪᒪᓚᐅᖅᑐᖅ, ᓯᓚᕐᑰᑕᐅᖃᑦᑕᖅᓗᓂ, ᐊᒻᒪᓗ ᒪᕿᐃᓐᓈᖕᓄᑦ ᐅᐸᓗᓂᑦ ᖃᕐᓗᑕᓚᐅᖅᑐᖅ. ᑕᐃᑉᓱᒪᓂ ᐱᕐᒪᕐᓇᖅᐸᓯᕐᒪᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒥᑦ ᓇᓐᓄᒐᒃᓴᖅᑲᕈᖖᑦᑎᐊᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ. ᑖᒃᓴᖃᓚᐅᕐᒋᑦᑐᒍᑦ/ᓇᓐᓄᒐᒃᓴᖅᑕᖃᓚᐅᕐᒋᑦᑐᒍᑦ ᐱᕐᒪᕐᓇᖅᐸᓯᕐᒪᓂᑦ ᑕᓯᐅᔭᕝᕙᐊᕐᒧᑦ ᑭᓯᐊᓂ 2010-ᖑᓕᕐᒪᑦ. ᐊᒃᓯᓪᓘᓐᓃᑦ 38-ᓂᒃ ᐱᖃᓕᕋᑉᑕ. ᑕᐃᑉᓱᒪᓂᐅᓚᐅᖅᑐᒧᐊᖅᑐᕐᓗ ᑎᑎᕋᐅᓯᐅᓂᐊᕐᒪᑦ. 2018-ᒥᑦ, ᑲᓇᕐᓇᒥᑦ ᐊᓄᓇᑐᔪᒪᓇᐅᖃᑦᑕᓚᐅᖅᑐᖅ, ᐊᒻᒪᓗ ᑕᓇᐅᒃᑯᑦ ᓇᓄᓗᒃᑖᑦ ᐅᐊᕐᓇᖅᐸᓯᕐᓗᓄᑦ ᑯᐸᒃ ᖃᓂᑖᓄᐊᓚᐅᖅᓯᒪᔪᑦ. ᓯᓚᑎᕗᑦ ᑕᐅᓇᓂ, ᐅᖃᓚᐅᖅᑐᑦ ᓈᓚᐅᑎᒃᑯᑦ ᐊᒥᓱᒪᓇᐊᓗᕐᓂᑦ ᓇᓄᖅᑕᖃᓕᕐᓂᕋᐃᐸᓗᑎᒃ ᖃᓂᑖᕐᓗᓂᑦ ᓴᓂᑭᓗᐊᑉ ᐊᒻᒪᓗ ᐅᐊᕐᓇᖅᐸᓯᕐᓗᓂᑦ ᑯᐸᒃ.
- **ᐊᓕᓴ**: ᒪᑦᓈᖅᐸᒋᑦ ᐋᓕᒃᓴ, ᐅᖃᐅᓯᕆᓚᐅᒃᑎᒃ ᑕᒪᒃᑯᐊ ᐅᕙᑎᖕᓄᑦ.
- **ᐊᕙᓐ**: ᐊᓕᐊᓇᐃᒋᔭᕋ ᖃᐅᔨᒪᔭᕆᐊᖃᕋᐸᑎᒍᑦ ᖃᓄᖅ ᐊᕙᑎ ᐱᑎᑦᑎᖃᑦᑕᕐᒪᖔᑦ ᓇᓄᐃᑦ ᓇᒨᙵᐅᓯᒪᓂᕐᒋᓐᓄᑐᐃᓐᓇᐅᕐᒋᑦᑐᖅ, ᑭᓯᐊᓂᑦᑕᐅᖅ ᐱᐊᓂᑎᐊᖅᓯᒪᔭᕆᐊᖃᕐᓂᕐᓚᑕ ᓈᓴᐃᖃᑦᑕᕐᓂᕗᑦ. ᑕᒃᓯᖅᑐᒻᒪᕆᐅᒃᐸᑦ, ᖃᕐᓗᑖᒃᑯᑦ ᓈᓴᐃᓗᓂ ᐱᐅᓗᐊᕐᒋᑑᔪᖅ. ᑕᒃᑯᐊᓗᒃᑖᖅ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᔭᕆᐊᖃᖅᑐᑦ ᐃᓱᒪᒋᑎᓪᓗᑎᒍᑦ ᖃᓄᐃᖕᓕᓂᐅᔪᑦ.

- **ᑭᐊᘄᓐ**: 2016-ᒥᑦ, 2021, 2022-ᒥᑦ ᐊᒥᓲᑎᒋᓂᕐᒋᓐᓂᑦ, ᓱᓇᐅᕙ ᐳᓴᓐᑎᕐᓕ ᓂᖁᕐᒋᓐᓂᒃ ᐆᒃᑐᕋᐃᕐᒪᑕ?
- **ᐊᕙᓐ**: ᐃᓱᒪᒋᔭᕋ ᖃᕆᑕᐅᔭᒃᑯᑦ ᑐᖅᖁᖅᓯᒪᔪᑦ, ᐊᒻᒪᓗ ᐃᓚᐅᔪᑦ ᓯᕗᓂᐊᒍᑦ ᐆᒃᑐᕋᐅᑕᐅᓂᐊᖅᑐᑦ ᑲᑎᑕᐅᔪᑦ ᓇᓄᕐᓂᑦ ᐊᒻᒪᓗ ᑰᖓᔪᐊᕐᒥᑦ ᐃᓚᒋᔭᐅᐸᓗᓂ ᓇᓄᐃᑦ ᒥᒃᓵᓄᑦ ᐅᔾᔨᖅᑐᖁᔨᓂᕐᒧᑦ ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑕᐃᑲᓂ ᐱᓕᕆᖃᑦᑕᖅᓱᑎᒃ, ᐃᓱᒪᔪᕐᓕ ᓂᖁᕐᒋᓐᓂᑦ ᐲᔭᐃᓚᐅᖅᑐᑦ ᖃᓂᑕᕐᓗᓂ 900 ᓇᓄᕐᓂᑦ, ᑭᓯᐊᓂ DNA-ᓂᑦ

ᓇᓗᓇᐃᒃᑯᑦᑲᖅᑐᒍᑦ ᑲᑎᓗᒃᑖᖅᑯᒋᑦ ᖃᓂᑖᓃᑦᑐᓂᒃ 3000 ᓇᕆᐃᑦ, ᑭᓯᐊᓂ ᓱᕐᓗ 1980-ᒥᑦ ᐱᒋᐊᖅᓯᒪᔪᑦ ᑕᒪᐅᖓ ᑎᑭᕐᕈᒍ. 40 ᐅᑭᐅᓅᔪᖅ ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ, ᐊᒥᓱᒻᒪᕆᐊᓗᐃᑦ ᓇᕆᐃᑦ ᐆᒪᙱᓕᖅᑐᑦ. ᑭᓯᐊᓂ ᓂᕿᖏᓐᓂᒃ ᐲᔭᐃᓂᕐᒧᑦ 900 ᖃᓂᑖᓂᑦ ᐆᒃᑐᕋᐅᑕᐅᓂᐊᖅᑐᓂᒃ ᐱᓚᐅᖅᓯᒪᔪᑦ.

- **ᑭᐊᕙᓐ:** ᐃᓕᑕᕆᔭᕗᑦ ᐱᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑕᓯᐅᔭᕐᒥᑦ ᓇᕆᐃᑦ ᓂᕿᖏᑦ ᐲᖅᓯᐸᓗᓯ ᑲᑎᑦᑎᖃᑦᑕᖅᑐᓯ?
- **ᐊᕙᓐ:** ᐋ, ᐊᑕᐅᓯᖅ ᐱᔾᔪᑕᐅᔪᖅ ᓇᓐᓄᒃᓯᒪᔭᕆᐊᖃᖅᑐᑎᑦ ᓇᓂᑭᖓ ᓇᓐᓄᒃᑕᕆᖓᓂ ᓯᕗᓪᓕᖅᐹᒥᑦ, ᖃᐅᔨᒪᙱᑦᑐᑎᑦ ᐱᓱᓂᕐᒪᖓᓛᑦ ᑕᓯᐅᔭᕐᒥᙵᓛᖅᑯᓂ ᑕᒪᐅᖓᖅᑐᖅ. ᐃᑲᔫᖑᔪᖅ ᖃᑉᓯᐊᖕᔪᖕᓂᑦ ᐆᒃᑐᕋᐅᑎᑖᕈᕕᑦ ᐊᑕᐅᓯᕐᒥᑦ ᓇᕆᕐᒥᑦ. ᑕᑯᓚᐅᖅᑕᕗᑦ ᓇᕆᖅ ᐅᐊᖕᓇᖓᓂᑦ ᑰᔪᐊᑉ, ᑕᑯᓕᓐᑉᓗᒍ ᓂᐅᓪᓴᓐ ᑰᖕᒥᑦ (Nelson River) ᐊᒻᒪᓗ ᐅᑎᓐᑉᓗᓂ ᐅᐊᖕᓇᖓᓄᑦ ᑰᔪᐊᑉ. ᐱᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᕆᖑᔪᖅ ᑕᑯᕐᑲᑦᑕᕋᑎᒍᑦ ᐱᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ, ᑭᓯᐊᓂ ᐃᓚᖏᑦ ᐅᑎᖅᑕᕐᑲᑦᑕᖅᑐᑦ. ᑕᒪᓇ ᖃᐅᔨᒪᓕᓐᐊᕐᑲᖅᑕᕗᑦ ᓇᓂ ᓇᕆᕐᓂᑦ ᑕᑯᓚᐅᖅᓯᒪᔭᑉᓯᖓᓂᑦ ᐊᒻᒪᓗ ᓇᓃᑦᑐᒪᕐᑲᑦᑕᕐᒪᖔᑕ, ᐊᒻᒪᓗ ᐊᔾᔨᒥᒃ ᑕᑯᔪᖓᖅᓯᓗᑎᑦ.

ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᖏᓪᓗᑎᒃ ᐱᓕᕆᐊᒃᓴᐃᑦ ᐱᖓᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ

- **ᐋᓕᒃᔅ:** ᐊᓕᐊᓇᐃᒍᓱᒃᑐᖓ ᑐᓵᑉᓗᖓ ᐃᓚᖏᓐᓂᒃ ᐊᔾᔨᒌᙱᑦᑐᓂᒃ ᐊᑐᖅᑕᐅᕐᑲᑦᑕᖅᑐᓂᒃ ᕿᒫᑎᑕᐅᕐᑲᑦᑕᕐᓂᖏᑦ ᓇᕆᐃᑦ, ᑐᓂᔭᐸᓯᖓᓂᑦ. ᑕᐅᑐᒃᑯᖓ ᓇᕆᕐᒥᑦ ᑰᔪᐊᕐᒥᑦ, ᑕᑯᓯᒪᔪᖓ ᓇᕆᕐᓂᑦ ᕿᒫᑎᑦᑎᓇᓱᒃᑯᑎᒃ ᓂᐱᖅᑯᖅᑐᔪᓂᒃ ᐊᑐᖅᑕᐅᓗᐊᕐᑲᑦᑕᖅᑐᓂᑦ. ᓇᕆᖅ ᕿᒫᑎᓪᓗᒍ, ᐊᑐᖅᑕᐅᖏᓐᓇᖅᑯᓂ ᓂᐱᖅᑯᖅᑐᔪᑦ ᐊᒻᒪᓗ ᓯᖅᑯᖅᑎᑖᖅᑕᐅᑉᓗᓂ ᓇᕆᖅ. ᓇᕆᐃᑦ ᑐᓵᑎᐊᖅᑐᒻᒪᕆᐊᓗᐃᑦ. ᐊᑐᕐᑲᑦᑕᖅᑕᖓᑦ ᓂᕆᔭᒃᓴᖅᓯᐅᖅᑯᑎᒃ, ᑐᓵᕐᑲᑦᑕᖅᑐᑦ ᓇᑦᑎᕐᓂᑦ ᓯᑯᑉ ᐊᑖᓃᑦᑐᓂᒃ. ᓇᐃᒪᔪᖕᓇᖅᑐᑦ 50 ᒪᐃᓕᓂᒃ ᐅᖓᓯᒃᑎᒋᔪᒥᑦ. ᖁᑭᐊᔭᐅᓗᐊᖅᐸᑕ, ᑐᓵᔪᖕᓃᕈᖕᓇᖅᑐᑦ. ᐊᕙᐅᕐᒧᐊᕌᖓᑕ, ᑐᓵᑦᑎᐊᕐᓂᖅ ᐊᔪᕐᑲᑦᑕᖅᑐᑦ. ᐆᒪᔪᓕᕆᔨᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ ᑕᐃᒪᓐᓇᐃᓕᐅᖅᑐᑦ, ᐊᒻᒪᓗ ᐊᔪᕈᖕᓃᖅᓴᖅᑕᐅᑦᑎᐊᕆᐊᕐᑲᖅᑐᑦ. ᐃᒻᒪᖄ ᐱᓕᕆᕝᕙᒋᔭᐃᑦ ᖃᓄᐃᓕᐅᕐᑯᔨᔪᖓᖅᑐᖅ.
- **ᐊᓕᓵ:** ᖁᕙᐊᒋᔭᕋ ᑐᓂᓯᒐᕙᑦ ᑕᒪᒃᑯᓂᙱᑦ ᓇᓗᓇᐃᕈᖓᕐᓂᐊᕋᑦᑖᓐᓃᑦ, ᓄᓇᕗᒻᒥᑦ, ᐊᑐᓗᐊᖅᑕᐅᑦᑕᐃᓕᑎᓪᓗᒋᑦ ᓂᐱᖅᑯᖅᑐᔪᑦ ᐊᑐᖅᑕᐅᕐᑲᑦᑕᖅᑐᑦ ᕿᒫᑎᑦᑎᓇᓱᒃᑯᑎᒃ. ᑭᓯᐊᓂ, ᒫᓂᑑᐸᒥᑦ ᐆᒪᔪᓕᕆᔩᑦ ᖃᓄᐃᓕᓂᖅ ᐊᔪᖅᑕᒃᑲ ᐅᕝᕙᓘᓐᓃᑦ ᐊᕙᓐ-ᒧᑦ ᐱᓕᕆᕝᕙᒋᙵᕆᓐᓇᐸᑎᒍᑦ. ᑐᓂᔾᔪᑎᒋᔪᖓᖅᑕᕗᑦ ᒫᓂᑑᐸᒦᑦᑐᓄᑦ ᐅᐸᓗᒥ ᐅᖃᐅᓯᕆᔭᓯ.
- **ᐊᕙᓐ:** ᒪᑦᓈᖅᐸᕆᑦ ᐋᓕᒃᔅ. ᐊᓯᖓ ᖃᐅᔨᔭᐅᖁᔭᕋ ᑲᑎᕐᒪᔪᓂᒃ ᐱᑕᖃᖅᑐᖅ ᑕᐃᔭᐅᔪᓂᒃ ᓇᕆᕐᓂᑦ ᐱᓕᕆᐊᖃᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᕋᓃᓂᒃ ᐅᑯᐊᖑᐸᓗᑎᒃ ᐅᑭᐅᖅᑕᖅᑐᒥᑦ ᐊᕙᒃᑐᖅᓯᒪᔪᒥᑦ ᐊᒻᒪᓗ ᑲᓇᑕᐅᑉ ᓄᓇᖓᑦ ᐊᕙᒃᑐᖅᓯᒪᔪᓂᑦ ᐆᒪᔪᓕᕆᔨᓂᑦ ᓇᕆᕐᓂᒃ ᖃᐅᔨᓴᐃᖃᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᐸᓗᑎᒃ. ᐊᕐᓇᖅᑕᖅᑐᖅ ᑲᑎᒪᔨᐊᑖᒥᑦ ᐃᓚᐅᔪᒥᑦ ᑭᒡᓚᖅᑐᐃᔪᖅ ᒫᓂᑑᐸᒥᑦ. ᐆᒪᔪᓕᕆᔨᐅᔪᖅ ᑖᓚᓴᒥᐅᑕᖅ. ᐊᑎᖓ ᕙᑭ ᑐᕆᓪ ᐊᒻᒪᓗ ᑲᒪᔩᑦᓚᐅᔪᖅ ᐊᐅᓚᑦᑎᓂᕐᒥᒃ ᓇᕆᕐᓂᑦ ᓇᕆᐃᑦ ᒥᒃᓵᓄᑦ ᐅᔾᔨᖅᑐᖁᔨᓂᕐᒧᑦ ᑐᑖᖓᔪᒥᑦ. ᑐᓂᔪᖓᑦᑎᐊᖅᑕᕗᑦ ᐅᖃᐅᓯᕆᔭᓯ ᕙᑭᒧᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ ᐊᑐᖅᑕᐅᕐᑲᑦᑕᕐᓂᖏᑦ ᓇᕆᕐᓂᒃ ᕿᒫᑎᑦᑎᓇᓱᒃᑐᑦ ᓂᐱᖅᑯᖅᑐᔭᐃᑦ ᐊᒻᒪᓗ ᐃᓱᒪᒎᑎᒋᔭᐅᔪᑦ ᓄᓇᓕᖕᒥᐅᓄᑦ ᓇᕆᐃᑦ ᑐᓵᑎᐊᕐᓂᖏᓐᓄᑦ ᐊᒻᒪᓗ ᐃᕙᐃᓗᑕᐅᔪᑦ ᓇᕆᐃᑦ ᓄᓇᓕᖕᒧᐊᕐᕕᖓᑕ.

- **ᐲᑕ:** ᑰᔪᐊᕐᒥᑦ, ᓂᕐᔪᓯᓇᓐᑦ ᓅᑎᔭᕐᕖᖓᑕ ᓇᕆᕐᓂᒃ ᖁᓕᒥᒌᓕᖕᓂᑦ ᐊᑐᖅᑲᑎᒃ, ᐅᐊᖕᓇᒧᑦ ᐊᒡᔭᖅᑕᐅᕐᑲᑦᑕᖅᑐᑦ ᑭᒡᓕᓕᐅᖅᓯᒪᔫᒃ ᖃᓂᑖᓄᑦ. ᐊᖕᒋᕋᕐᑎᖓᓂᑦ, ᐃᓄᖕᓄᑦ ᐱᐅᕆᔭᐅᖕᒋᑦᑐᖅ. ᓱᕐᓕ ᐅᑎᖅᑎᑕᐅᕐᑲᑦᑕᙱᓕᖅ ᓇᕆᖅ ᐱᓚᐅᖅᑕᖕᒋᓐᓄᑦ?
- **ᐊᕙᓐ:** ᐱᔾᔪᑕᐅᔪᖅ ᐅᖃᐅᓯᐅᕆᔪᒃᑐᖅ. ᐅᖃᓪᓚᒃᑯᕙᑦ ᒫᓂᑑᐸᒧᑦ, ᐱᖅᑯᓯᕆᖓᒪᒋᑦ ᓇᕆᐃᑦ ᐅᐊᖕᓇᒧᑦ ᐅᑎᕐᓗᑎᒃ, ᐱᓱᓕᖅᑲᑦᑕᖅᑐᑦ ᐅᐊᖕᓇᒧᑦ, ᑕᓇᐅᑉ ᓯᑯᖓᓄᐊᕐᓂᐊᕐᓗᑎᒃ ᐅᑭᐅᒃᓴᒃᑯᑦ, ᑕᒪᓐᓇ ᐱᔾᔪᑎᒋᔭᖓᑦ. ᐅᖃᙱᑦᑐᖓ ᓇᓚᐅᑦᑑᖓᒪᑕ, ᑭᓯᐊᓂ ᐱᔾᔪᑎᒋᔭᖓᑦ ᖃᖓᑕᓱᒃᑯᑦ ᓂᒋᐊᓄᙱᑦᑕᐅᑎᑦᑕᙱᕆᓐᓂᖏᕆᓐᓄᑦ ᐅᑎᒧᑦ ᐱᓱᖕᓂᐊᕐᒪᑕ ᐊᒻᒪᓗ ᐃᓅᐃᓗᑕᐅᒃᑲᓐᓂᕐᓗᓂ ᑰᔪᐊᕐᒧᑦ. ᑕᐃᒪᓐᓇᒧᑦ ᐊᑭᐅᓄᑦ ᑰᑉ ᐊᒡᔭᖅᑐᐊᕐᑲᑦᑕᖅᑐᑦ, ᑭᓯᐊᓂ ᓇᓗᓇᙱᑦᑐᖅ ᖃᐅᔨᒪᒐᑉᓯ, ᐱᓱᕐᑲᑦᑕᖅᑐᑦ ᑕᒃᐸᐅᖓ ᓯᒡᔭᒃᑯᑦ, ᐊᒻᒪᓗ ᐊᕐᕙᐊᓄᐊᕐᓗᑎᒃ ᐃᓅᐃᓗᑕᐅᓕᕐᓗᑎᒃ.
- ᐃᓱᒪᔪᖓ ᐱᑦᑕᐅᓇᔭᖅᑐᖅ ᑭᑐᓗᒃᑖᑦ ᖃᓇᐅᕐᒥᑦᑐᑦ ᐊᑕᐅᑦᑎᒃᑯᑦ, ᓱᕐᓗ ᒫᓂᑑᐸᑐᑦ, ᐆᒪᔪᓕᕆᔨᐊᑎᒃᑯᓗᒃᑖᖅ, ᐊᒻᒪᓗ ᐃᒻᒪᖄ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐅᖃᐅᓯᖃᕐᓗᑎᒃ ᐃᓂᐅᔪᒥᑦ ᐱᔪᓇᕐᓂᖃᕐᓂᐅᔪᓄᑦ ᑐᓗᖅᑕᕈᑕᐅᔪᓂᒃ.

- **ᐋᓕᒃᔅ:** ᐱᖃᓐᓈᖃᖅᑐᖓ ᓴᓪᓕᕐᒥᑦ ᑕᑯᓯᒪᔪᓂᒃ ᓇᕆᕐᓂᑦ ᑐᕌᔪᐊᖓᔪᒥᑦ ᒥᕈᐊᖅᑕᐅᓯᒪᔪᓂᒃ ᑐᓄᐊᒍᑦ, ᒥᕈᐊᖅᑕᐅᓚᐅᖅᑐᓂᒃ ᑰᔪᐊᕐᒥᑦ. ᓴᓪᓕᕐᒨᙵᐅᔪᖕᓇᖅᑐᑦ, ᑕᒪᓇ ᐃᓕᑉᓯᖕᓄᑦ ᖃᐅᔨᔭᐅᖁᖅᑲᐅᔭᕋ.

- **ᑭᐊᕙᓐ:** ᐊᕐᕙᐊᓂᑦ, ᐊᒥᕋᐃᔭᕐᕙᖕᒥᑦ, ᐅᑭᐅᑖᓪ-ᓗᐊᒥᑦ ᐊᒻᒪᓗ ᐃᓚᖏᑦ ᑲᑖᒪᓇᕝᕙᖕᒥᑦ, ᑕᑯᖅᑲᑦᑕᖅᑐᒍᑦ ᐊᒥᓱᓂᒃ ᓇᕆᕐᓂᑦ ᐊᒃᑕᑯᓂᑦ. ᐃᓖᓐᓂᒃᑯᑦ ᑕᑯᖅᑲᑦᑕᖅᑐᒍᑦ ᐊᖏᔪᒥᒃ ᓇᕆᕐᒥᑦ ᓴᓪᓗᐊᓘᑉᓗᓂ ᑭᖑᓪᓖᖕᒋᑦ ᐊᑐᕐᓗᒋᑦ ᐱᓱᖕᓂᖅ ᐊᔪᕐᓚᑦᑎᐊᖅᑐᒥᑦ. ᓇᕆᐃᑦ ᓴᐅᓃᓐᐅᔪᑦ ᐊᒻᒪᓗ ᐅᓗᕆᐊᓇᖅᑐᒻᒪᕆᐅᔪᑦ. ᓂᕿᖏᓐᓂᒃ ᐲᖅᓯᕝᕙᐅᖅᑲᑦᑕᖅᐸᑕ ᒥᒥᐊᓂᑦ, ᐋᓐᓂᐊᕐᓂᖅᑖᕈᓐᓇᖅᐸ?
- **ᐊᕙᓐ:** ᑲᐳᑏᑦ ᐊᑐᖅᑲᑦᑕᖅᑕᕗᑦ ᓴᓗᒪᖅᑕᐅᓯᒪᔪᑦ ᐋᓐᓂᐊᕐᓇᕐᓂᑦ ᑲᑎᓯᒪᐸᓗᑎᒃ ᑎᑭᒃᑳᖓᓚ. ᐱᑕᖃᕈᓇᐃᓐᓇᓱᑦ, ᐲᕓᖓᐸᑎᒍᑦ ᕗᕆᓐᓂᑦ. Hᐊᓐᓇᕆᓐᓃᑦᑐᓂᒃ ᓂᕿᖏᓐᓂᑦ ᐲᖅᓯᖃᑦᑕᖅᓯᒪᔪᒍᑦ, ᑕᐃᒪᓐᓇᐃᓕᐅᖃᑦᑕᕋᔪᖓᑕ, ᐊᒥᓱᓂᒃ ᑕᑯᖃᑦᑕᕋᔭᖅᑐᒍᑦ, ᑕᐃᒪᓐᓇᒧᑦ ᐋᓐᓂᐊᕐᓇᖅᑖᕈᑕᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐃᓱᒪᙱᑦᑐᒍᑦ.

- **ᑭᐊᕙᓐ:** ᑕᑯᖃᑦᑕᓕᓵᓚᐅᕋᑉᑕ ᑲᑉᐱᐊᓇᑦᑎᐊᖅᑐᓂᒃ ᓴᓪᓗᒻᒪᕆᐊᓗᖕᓂᑦ ᓇᓄᕐᓂᑦ. ᑕᑯᔭᐅᓴᔪᓚᐅᖅᓯᒪᙱᑦᑐᑦ ᐅᑭᐅᒃᓵᒃᑯᑦ ᐅᑎᖅᑕᖅᑐᓂᑦ.
- **ᐊᓕᓵ:** ᖁᔭᓐᓇᒦᒃ. ᐃᓱᒪᓇᓱᒃᑐᖓ ᓱᓇᒃᑲᓐᓂᐅᓇᔭᕐᒪᖔᑦ. ᖃᐅᔨᒪᔪᖓ ᐃᓛᓐᓂᒃᑯᑦ ᐋᓐᓂᕐᓂᖃᕈᖕᓇᖅᑐᑦ ᐊᑐᖅᑕᐅᓂᖕᒋᑦ ᐊᒥᓱᑐᐊᑦ ᕈᐊᐳ ᖃᕋᐃᑦ ᑕᐃᒪᓐᓇᐃᑦᑐᕋᓗᐃᑦᑑᓐᓃᑦᑕᐅᖅ. ᑕᑯᓯᒪᔪᖓ ᕿᕐᓂᖅᑐᓂᑦ ᓇᓄᕐᓂᑦ ᐊᒻᒪᓗ ᐃᓱᒪᓇᓱᒃᑐᖓ ᐊᓯᖕᒋᓐᓂᒃ ᐱᔾᔪᑕᐅᔪᓂᒃ. ᐱᑦᑕᐅᔪᖅ ᑕᑯᓯᒪᔭᐃᑦ. ᐃᓱᒪᔪᖓ ᑕᒪᓐᓇ ᐃᓱᒪᑉᑎᖕᓃᑎᓪᓗᒍ.
- **ᐋᓕᒃᓴ:** ᑕᑯᓯᒪᔪᖓ ᓇᖕᒥᓂᖅ, ᑖᕝᓂ ᐊᕐᕕᐊᓂᑦ, ᑐᖁᑦᑎᔭᕆᐊᖅᑲᖅᑕᖅᓯᒪᒪᑉᑕ ᓇᓄᕐᓂᑦ ᓴᓪᓘᓗᐊᕐᓂᖕᒋᓐᓄᑦ. ᓇᓂᓯᓯᒪᔪᒍᑦ ᕈᐊᐳ ᖃᕐᔪᕐᓂᑦ ᓂᕿᐊᓃᑦᑐᓂᒃ ᑐᖁᑕᐅᔭᕌᖓᑕ.
- **ᐊᓕᓵ:** ᐱᓪᒪᕆᐅᔪᖅ ᑕᐃᒪᓐᓇᐃᑦᑐᒥᒃ ᑕᑯᒃᑲᓐᓂᕈᐸᓯ, ᐊᔾᔨᓕᐅᕐᓗᒋᑦ ᐊᒻᒪᓗ ᐅᕙᑉᑎᖕᓄᑦ ᐊᐅᓪᓚᖅᑎᓪᓗᒋᑦ ᓇᓄᓕᕆᕝᕙᖕᒍᑦ ᐅᕝᕙᓘᓐᓃᑦ ᔫ ᓄᑲᖅᑳᕐᒍᐊᕐᓗᒋᑦ. ᖃᐅᔨᓴᕈᒪᔭᕗᑦ, ᐃᓗᐃᓗᑕᙱᖅᐸᓪᓚᐊᓂᐊᖕᒋᓪᓗᓂ ᐊᒻᒪᓗ ᐱᑎᑦᑎᓂᖃᕐᓂᐊᖕᒋᓪᓗᑎᒃ ᐃᓗᐃᓗᑕᙱᕐᓂᖕᒋᓐᓄᑦ ᓇᓄᐃᑦ.
- **ᐋᓕᒃᓴ:** ᔫᑉ ᖃᐅᔨᒪᓇᔭᖅᑐᖅ ᑕᐃᑲᓃᓚᐅᕐᒪᑦ ᑕᑯᑉᐳᓂᐅᒃ.
- **ᐊᓕᓵ:** ᒪᑦᓈᖅᐸᕆᑦ ᐋᓕᒃᓴ, ᖃᐅᔨᕆᐊᕐᕙᕆᓂᐊᖅᐸᕋ.

ᑲᑎᒍᒃᑖᖅᑯᕆᑦ ᓇᓄᒃᑕᐅᔭᖕᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ ᐅᕐᑲᓪᓗᓇᒃᓴᐅᑎᓃᑦ

- **ᐋᓕᒃᓴ:** ᐅᖃᐅᓯᐅᔭᖅ ᐃᒃᐱᖕᓇᖅᑐᒻᒪᕆᐅᔭᖕᓇᖅᑐᖅ ᐊᒻᒪᓗ ᐊᒥᓱᓄᑦ ᐅᐸᓗᓄᑦ ᐅᖃᐅᓯᕆᓗᒍ ᐊᒻᒪᓗ ᖃᓄᑐᐃᓐᓇᖅ ᖃᓄᐃᓕᐅᖁᔨᓂᕐᒥᑦ ᐱᙱᓪᓗᑕ. ᑭᓯᐊᓂ, ᑲᑎᒍᒃᑖᖅᑯᕆᑦ ᓇᓄᒃᑕᐅᔭᖕᓇᖅᑐᑦ, ᐊᑯᓐᓂᖕᒋᓐᓂᑦ ᓄᓇᓕᖕᓂᑦ, ᐱᓕᕆᐊᕆᖃᑦᑕᖅᑕᑉᑎᖕᓂᑦ, ᐅᓇᑕᕈᑎᕆᖃᑦᑕᖅᑕᑉᑎᖕᓂᑦ, ᐱᒡᒎᒍᑎᕆᖃᑦᑕᖅᑕᑉᑎᖕᓂᑦ, ᐊᑐᖅᐸᒃᑕᕞᖕᒋᑦᑐᖅ ᐊᕐᕕᐊᓂᑦ, ᑎᑭᕋᕐᔪᐊᕐᒥᑦ, ᐊᒻᒪᓗ ᑲᖕᒋᖅᓵᓂᕐᒥᑦ. ᐅᓇᑕᕈᑎᖃᑦᑕᖅᑐᒍᑦ ᐊᒥᓲᓛᖑᕆᐊᖅᑐᓂᒃ ᐱᓂᐊᕐᓗᑕ ᑲᑎᒍᒃᑖᖅᑯᕆᑦ ᓇᓄᒃᑕᐅᔭᖕᓇᖅᑐᓂᑦ. ᓇᓄᖅᓯᐅᕐᓂᖅ ᐃᓄᐃᑦ ᐱᖁᓯᕆᖕᒪᒍ ᒪᖃᐃᑎᙵᕐᓂᐊᕐᓗᓂ. ᑕᑯᓚᐅᖅᓯᒪᒍᖕᓂ ᑖᕆᔭᐅᑎ Big Game, ᐊᖕᒍᑏᑦ ᐊᖕᒍᑎᙵᖅᓱᑎᒃ ᓯᕗᓪᓕᖅᐸᑦᑲᒥᒃ ᓂᕐᔪᑎᒥᒃ ᐱᑕᖅᐸᑕ, ᐅᕙᒍᑦ ᐃᓄᒃᑎᒍᑦ ᐱᖅᑯᓯᕆᔭᕗᑦ, ᖃᓄᖅ ᓄᑲᑉᐱᐊᖅ ᐊᖕᒍᑎᙵᕐᑲᑦᑕᕐᓂᖓ, ᒪᖃᐃᑎᙵᖅᓗᓂ. ᑕᐅᑐᕆᐊᒃᑲᑕᐃᑦ.
- ᐱᖓᓱᓂᒃ ᐃᕐᓂᖃᖅᑐᖓ. ᐊᖓᔪᒃᓕᖅᐹᖅ 30-ᓂᒃ ᐅᑭᐅᖅᑖᕐᓂᐊᓕᖅᑐᖅ. ᐃᓅᓱᒃᑖᖓᓂᑦ ᓇᓄᕐᒥᑦ ᓇᓂᓯᓚᐅᖅᓯᒪᙱᑦᑐᖅ. ᑐᒡᓕᐊ 20-ᓂᒃ ᐅᑭᐅᖅᑲᖅᑐᖅ ᖁᑭᖅᓯᓚᐅᖅᓯᒪᙱᑦᑐᖅ ᓇᓄᕐᒥᑦ. ᐱᖓᓱᖓ, 16-ᓂᒃ ᐅᑭᐅᖅᑲᖅᑐᖅ, ᖁᑭᖅᓯᔪᖕᓇᖅᓯᓚᐅᖅᓯᒪᙱᑦᑐᖅ ᓇᓄᕐᓂᑦ ᐱᔾᔪᑎᕆᐸᓗᒍ ᑲᑎᒍᒃᑖᖅᑯᕆᑦ ᓇᓄᒃᑕᐅᔭᖕᓇᖅᑐᑦ ᐊᒥᓲᖕᒋᓗᐊᓚᐅᕐᒪᑕ. ᐅᕙᖓ, ᐊᑎᕋ ᐊᒨᔭᐅᓚᐅᖅᓯᒪᔪᖅ ᐊᑕᐅᓰᖅᓱᓂ ᐃᓅᓱᒃᑖᓪᓂᑦ. ᖃᐅᔨᒪᕐᑯᔨᑐᐃᓐᓇᖅᑐᖓ ᐃᓕᐸᓯᖕᓂᑦ ᐱᓪᓚᕆᐅᑎᒋᓂᖓᑕ ᑐᓂᓗᒍ ᓄᓇᓕᒃ ᑲᑎᒍᒃᑖᖅᑯᕆᑦ ᓇᓄᒃᑕᐅᔭᖕᓇᖅᑐᓂᑦ, ᐱᖅᑯᓯᕗᑦ ᐱᔾᔪᑎᕆᐸᓗᒍ.
- **ᐊᕙᓐ:** ᒪᑦᓇ ᐋᓕᒃᓴ ᐅᖃᐅᓯᕆᓴᖕᓄᑦ. ᐱᓪᒪᕆᐅᔪᖅ ᑕᒪᐃᓐᓇᐃᑦᑐᓂᒃ ᑐᓵᓗᓂ ᐃᑲᔪᕐᓂᐊᕐᓗᓂ ᐋᖅᑭᒃᑕᐅᓂᐊᖅᑐᓂᒃ ᐱᓂᐊᕐᓗᓂ ᐅᖃᐅᓯᕆᓴᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐱᓪᒪᕆᐅᓂᖓᑕ ᐱᖅᑯᓯᒃᑯᑦ ᐱᓪᒪᕆᐅᕆᔭᐅᓂᖕᒋᑦ ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᔪᑦ ᐊᒻᒪᓗ ᐊᖕᒋᓂᖅᓴᐅᔪᖅ ᐊᓯᐅᔾᑕᐃᓕᓂᕐᒧᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑕᒪᒃᑯᐊᓗᒃᑖᖅ ᐃᓱᒪᒋᔭᐅᔭᕆᐊᖃᖅᑐᑦ.
- **ᓗᐊᖕ:** ᐅᓂᑉᑳᕈᒪᔪᖓ ᐊᕐᖏᓂ ᐱᓚᐅᖅᑐᒥᑦ, ᖁᖓᑦᑐᓂᒃ ᑕᑯᔨᒪᑐᐃᓐᓇᖅᑐᖓ ᐳᐃᒍᕐᓂᐊᖕᒋᓐᓇᑉᑎᒍᑦ ᑕᒪᓐᓇ. ᐃᓄᖕᓂᑦ ᐊᕐᕕᐊᓄᑦ ᓈᓴᖅᑐᖅᑲᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᐃᓘᑦ ᑕᑯᓇᓯᕆᐸᓗᒃ ᐊᒪᕈᕐᒥᑦ ᓇᓂᕿᐊᖅ ᓄᓇᓕᐅᑉ ᐃᓗᐊᓂᑦ. ᐊᒦᔭᖅᓱᒍ ᐊᒻᒪᓗ ᕿᒻᒥᐅᓚᐅᖅᑐᖅ, ᑭᓯᐊᓂ ᐊᒪᕉᓇᓯᕆᔭᐅᓚᐅᖅᑐᖅ. ᖃᐅᔨᒪᓚᐅᕐᒥᑦᑐᑦ ᕿᖕᒥᐅᖕᒪᑦ, ᕿᒧᒃᓯᖅᑎᒧᑦ ᕿᖕᒥᕆᓴᐅᔪᖅ. ᒪᑦᓇ ᐅᓐᓅᔪᒧᑦ ᑐᓂᓯᑎᓪᓗᓯ ᓇᓄᐃᑦ ᒥᒃᓵᓄᑦ ᐊᒻᒪᓗ ᑕᐃᒪᑐᖅ ᑲᑎᒍᒃᑖᖅᑯᕆᑦ ᓇᓄᒃᑕᐅᔭᖕᓇᖅᑐᑦ ᐊᒥᓲᓂᖅᓴᐅᓂᐊᖅᐳᑦ ᐅᑭᐅᖅ ᐅᕝᕙᓘᓐᓃᑦ ᐊᒥᓲᓂᖅᓴᐅᓕᕐᓂᐊᖅᐳᑦ ᐅᑭᐅᓂᑦ ᖃᐃᔪᓂᑦ.
- **ᐲᑕ:** ᑕᒪᓐᓇ ᐱᓕᕆᐊᖅ ᐅᑭᐅᑕᒫᑦ ᐱᓕᕆᐊᕆᖅᑲᖅᑕᖅᐱᐅᒃ? ᖃᑉᐲᖅᑕᖅᐸᓱ ᑖᕋᙵᐅᖅᑲᖅᑕᖅᐱᓯ ᐊᒻᒪᓗ ᐅᓂᑉᑳᓕᐅᖅᐸᓱ ᑕᒪᑐᒥᙶᑦ?
- **ᐊᓕᓵ:** ᓄᓇᕗᓪᒥ ᒐᕙᒪᒃᑯᖑᐸᓗᑕ, ᓈᓴᐃᓇᓯᖅᑲᖅᑕᖅᑐᒍᑦ ᑖᓕᒪᐃᑦ ᐅᑭᐅᑖᒃᑯᑦ, ᐊᔪᙱᒃᑳᖓᐸᑕ ᐊᒻᒪᓗ ᐱᓂᐊᕐᓗᑕᑐᐃᓐᓇᖅ ᐃᒥᕐᑏᑎᕆᓂᖕᒋᓐᓂᒃ ᓇᓚᐅᑦᑖᖅᓯᒪᔪᒥᒃ, ᑭᓯᐊᓕ ᐊᕙᓐ ᐅᖃᓪᓚᒃᑎᓐᓂᐊᖅᑕᕋ ᐱᓕᕆᐊᖓᓄᑦ ᐊᒻᒪᓗ ᖃᓄᑎᕆᐅᔪᒥᒃ ᐱᓕᕆᖃᑦᑕᕐᓂᖕᒋᓐᓂᑦ.
- **ᐊᕙᓐ:** ᑕᑯᔭᖅᑐᐃᖃᑦᑕᓚᐅᖅᑐᒍᑦ ᓄᓇᓕᖕᓂᑦ ᐊᒻᒪᓗ ᖃᖓᑦᑎᐊᓵᖅ ᓯᕗᓕᖅᑎᐅᑉ ᐃᓂᐊᓄᐊᓚᐅᖅᑐᖓ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᑐᕋᖓᔪᓂᑦ ᑰᔪᐊᑉ ᖃᓂᑖᓂᑦ. ᑐᕋᖓᓇᔭᕗᑦ ᓯᕗᓪᓘᖕᓂᐊᕐᓗᑕ ᐅᑎᕝᕕᖅᑲᑦᑕᕐᓂᖅᓴᐅᓗᑦ ᓄᓇᓖᑦ ᐊᒻᒪᓗ ᑕᑯᔭᖅᑐᕐᓗᓯ ᐊᒻᒪᓗ ᐅᖃᓪᓚᖅᑲᑎᕆᓗᓯ ᑐᓴᓗᓯᓗ ᑕᑯᖃᑦᑕᖅᑕᑉᐱᓯᖕᓂᑦ ᐊᒻᒪᓗ ᖃᓄᐃᓕᐅᕐᒪᖔᑦ. ᑐᓂᓯᖃᑦᑕᕐᓂᖅᓴᐅᓗᑕ ᖃᓄᐃᖓᔪᑉᑎᖕᓂᑦ ᐱᓕᕆᐊᑉᑎᖕᓂᑦ ᐱᔪᓂᒃ ᐊᒻᒪᓗ ᐱᓕᕆᐊᕐᒥᑦ ᐱᓕᕆᑦᑎᐊᕐᓂᖅᓴᐅᓗᑕ ᐊᐅᓪᓚᖅᑎᑦᑎᖅᑲᑦᑕᕐᓂᕐᒥᐅᑕᐅᓇᐅᖕᒋᑦᑐᖅ ᓇᐃᑦᑐᓂᒃ ᐅᓂᑉᑲᓂᒃ, ᑎᑎᖅᑲᐅᑎᓂᒃ ᐃᓄᒃᑎᑐᓕᖅᑕᐅᓯᒪᔪᓂᒃ ᐊᒻᒪᓗ ᐱᓕᕆᑦᑎᐊᕐᓂᖅᓴᐅᓗᑕ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᑐᓂᓯᖅᑲᖅᑕᐅᑎᓂᕐᒧᑦ, ᑕᒪᑉᑕ ᑐᑭᓯᐊᓂᖅᑲᕈᓇᕐᓂᐊᕋᑉᑕ, ᐊᒻᒪᓗ ᐱᓕᕆᖅᑲᑎᒌᒡᓗᑕ ᓯᕗᓂᒃᓴᒥᑦ ᑭᑐᒃᑖᑦ ᓇᓄᖅᑕᖅᑎᓐᓂᐊᕐᓗᒍ.
- **ᑭᐊᕙᓐ:** ᐅᖃᐅᓯᖃᕈᒪᔪᖓ ᐊᑐᖅᓯᒪᔭᓪᓂᑦ ᐃᓅᓯᓪᓂᑦ ᐊᕐᕕᐊᓂᑦ. ᓄᑕᕋᐅᑉᓗᖓ ᐊᐅᔭᒃᑯᑦ, ᐊᑖᑕᒋᓚᐅᖅᑕᕋ ᐅᒥᐊᖃᓚᐅᕐᒪᑦ, ᐊᒻᒪᓗ ᐱᔪᓐᓇᑐᐊᕌᖓᑉᑕ, ᓯᐊᓐᑐᓇ ᕿᑭᖅᑕᓘᙵᐅᖃᑦᑕᖅᓱᑕ, ᕿᑭᖅᑕᑐᐊᕐᒍᖓᒪᑦ ᑕᓯᐅᔭᖅᔪᐊᕐᒥᑦ –

ᐱᐅᔪᒥᒃ ᓯᐅᕋᐅᓂᕐᒥᑦ ᓯᒡᔭᖅᑲᖅᓗᓂ ᐱᖓᓐᓇᖓᓃᑦᑐᒥᑦ ᐊᒻᒪᓗ ᑕᐅᑐᒃᑲᓐᒥᑦ ᓇᐸᔪᖅᑲᖅᓗᓂ ᐃᓄᓐᓄᑦ ᕿᙵᖕᒥᒋᐊᖅᑐᕐᕕᒋᔪᓐᓇᖅᓗᒍ ᑕᑯᔪᒪᑉᓗᑎᒃ ᑕᕆᐅᕐᒥᐅᑕᓂᒃ. ᐊᒥᓱᒪᕆᐊᓗᐃᑦ ᐃᒥᖅᖁᑕᐃᓚᐃᑦ ᐊᒻᒪᓗ ᒥᑏᑦ ᒪᓂᖕᒋᑦ. ᐅᒥᐊᑦ ᓄᓇᓕᑐᐊᕌᖕᒐᑕ, ᓄᑕᕋᐅᔪᒍᑦ ᐅᒥᐊᕐᒥᑦ ᐲᖅᓗᑕ ᐊᒻᒪᓗ ᖃᖅᑲᐅᑉ ᖄᖓᓄᙵᖅᑲᐅᑕᐅᓇᓱᒃᐸᒃᓗᑕ ᐊᒻᒪᓗ ᓯᕗᓪᓕᐅᓗᑕ ᐱᑉᐅᑕᕐᓂᐊᕐᓗᑕ ᐅᕙᓃᓐᓂᑦ ᓯᕗᓪᓕᐅᓗᑕ ᖄᖓᓄᐊᖅᑲᐅᑕᐅᓂᐊᕐᓗᑕ ᐃᓄᖕᒍᑦ ᓴᓇᔭᐅᓯᒪᔪᖅ ᓇᐸᔪᒍᑦ, ᐃᓱᒪᓗᖕᓇᖅᑐᖃᙵᕐᓗᓂ. ᐊᓈᓇᕗᑦ ᐅᕙᓃᓐᓂᑦ ᐊᑖᑕᕗᑦ ᐃᓯᒪᔪᖕᓇᓚᐅᖕᒋᑦᑐᓪᓘᓐᓃᑦ ᑕᓪᓚᑲᓗᐊᕈᑕ ᓇᒧᑉᑭᐊᖅ ᕿᑭᖅᑕᒥᑦ. ᑕᐃᒪᓐᓇ ᐃᓅᓯᐅᕙᓚᐅᖅᑐᖅ. ᐊᓈᓇᐅᔪᑦ ᐊᒻᒪᓗ ᐊᑖᑕᐅᔪᑦ ᐅᕙᓐᓂᓐᓃᑦ ᖃᑕᙳᑎᕆᑦ ᐱᑭᐅᑕᖅᑎᓪᓗᒍᑦ, ᖁᑭᐅᑎᓕᔭᖃᑦᑕᓚᐅᖅᑎᑦᑐᒍᓪᓘᓐᓃᑦ. ᑕᒪᐃᓐᓇ ᐃᓱᓖᓗᖕᓂᖅᖃᖅᐸᓚᐅᖅᑎᑦᑐᖅ ᐃᓅᓯᕐᒥᑦ.

- ᐅᑉᓗᒥ ᑕᐃᑯᙵᐅᒍᕕᑦ, ᕿᑭᖅᑕᒧᐊᕈᒪᙱᑦᑐᑎᓪᓗᓐᓃᑦ ᑦᑕᓚᒪᓂᒃ, ᐊᕝᕙᓂᓚᖕᓂᑦ, 7-ᓂᑦ, 8-ᓂᓪᓘᓐᓃᑦ ᓇᓄᖅᑕᕐᖂᓂᖕᒪᑦ ᕿᑭᖅᑕᒥᑦ. ᒪᑎᑕᐅᑎᕐᑲᕈᕕᑦ, ᐅᑕᖅᕿᓂᐊᖅᑐᑦ ᓯᒡᔭᒥᑦ ᓱᓇᑐᐃᓐᓇᒥᑦ ᐊᐅᓚᓂᖅᑲᖅᑐᒥᑦ. ᐃᖅᑲᓗᖕᒥᑦ ᒪᑎᑕᐅᓂᐊᖅᑐᖅᑲᖅᐸᑦ, ᒪᑎᑕᐅᑎᓄᐊᑲᐅᑎᒋᓗᑎᒃ ᐊᒻᒪᓗ ᓴᙱᓂᖕᒋᓐᓄᑦ, ᒪᑎᔮᖕᒋᑦᑐᑦ ᒪᑎᑕᐅᑎᒍᑦ, ᑮᓗᒍ ᐃᖅᑲᓗᒃ ᐲᕐᓗᒍᓗ. ᓇᑎᕈᕕᑦ ᐅᕝᕙᓘᓐᓃᑦ ᕿᓇᓗᒐᕈᕕᑦ, ᐃᓚᒋᔭᐅᓗᐊᖅᑐᑦ ᕿᓂᒋᔭᑉᑎᖕᓄᑦ, ᐱᓕᕆᐊᕆᔪᒪᙱᑦᑕᕗᑦ ᕿᑭᖅᑕᒥᑦ. ᐊᖏᕐᕋᐅᑎᓇᓱᒡᓗᒍ ᐅᕝᕙᓘᓐᓃᑦ ᐱᓕᕆᐊᕆᓇᓱᒡᓗᒍ ᐃᒻᒪᖄ ᐃᒫᓂᑦ ᓄᓇᒧᐊᕈᑕ, ᖃᐅᔨᒪᔪᒍᑦ ᓇᓄᕐᒥᑦ ᖃᐃᔪᖃᕐᓂᐊᕐᒪᑦ ᐊᒻᒪᓗ ᖃᑉᓯᐅᓗᑎᒃ ᖃᐃᓗᑎᒃ.
- ᐊᒥᓱᓄᑦ ᐅᑭᐅᓄᑦ, ᐋᓕᒃᓴ ᐅᖃᖅᖃᐅᓂᖓᑐᑦ, 30-40 ᐅᑭᐅᓄᑦ 20-ᓂᒃ ᓇᓄᒍᑎᓂᒃ ᐅᑎᖅᑎᕝᕕᐅᔪᒪᕐᑲᑦᑕᖅᓯᒪᔪᒍᑦ, ᓄᖅᑲᖅᑎᑕᐅᑦᑕᐃᓐᓇᓚᐅᖅᑐᓂᒃ 2007-ᒥᑦ ᐊᑯᓂᐅᔪᒧᑦ. ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᔪᒥᑦ ᐅᑕᖅᕿᓂᐊᖅᐱᑕ? ᖃᑯᒍᑦ ᐃᒡᓗᕈᖕᓚᕐᓂᐊᖅᐸ? ᖃᑉᔫᖅᑕᕐᓗᑕ ᑖᓐᓇᑦᑕᐃᓂᖅ ᐅᖃᐅᓯᕆᓚᐅᖅᑕᕗᑦ ᐅᖃᐅᓯᕆᔭᕆᐊᖃᖅᐱᑎᒍᑦ? ᖃᑯᒍ ᑐᔅᓴᖅᑕᐅᓕᕈᓖᖅᐸ? ᓱᓇᒥᒃ ᖃᓄᐃᓕᓂᐅᔪᒥᒃ ᐊᑐᕆᐊᖃᕐᓂᐊᖅᐱᑕ ᓇᓄᒍᑎᒃᓴᖅᑖᑕᐃᓇᕈᑎ ᐅᕙᓂᑦ ᐊᕙᑎᑦ ᐅᖓᑖᓃᖅᑐᓂᒃ? ᐊᑯᓂᓗᐊᖅ ᐅᑕᖅᕿᓕᖅᑐᒍᑦ. ᖃᓄᒃᑲᓂᖅ ᐅᖃᕆᐊᖃᖅᐱᑕ? ᐅᖃᕆᐊᖃᕋᓗᐊᖅᐱᑕᓘᓐᓃᑦ? ᐃᓱᒫᓘᑎᒋᔭᕗᑦ ᐅᖃᐅᓯᕆᓗᒋᑦ ᐃᑲᔪᕋᓗᐊᖅᐸ? ᓱᓇᒧᑦ ᐊᕘᑎᖃᖅᐸ ᑲᑎᒪᑎᑦᑎᓂᖅ ᐊᓯᖕᒋᓪᓘᓐᓃᑦ ᐊᔾᔨᑦᓚᖕᒋᓐᓄᑦ ᑲᑎᒪᑎᑦᑎᓂᐅᔪᑦ? ᖃᓄᖅ ᐃᒃᐱᒍᓱᒃᑲᖏᙱᓐᓂᖅᑲᖕᑲᕆᔭᓇᐊᖅᑲᖅᐱᑎᒍᑦ ᐃᓐᓇᑐᔫᑉ ᐊᒻᒪᓗ ᐱᖅᑯᓯᑐᔭᖕᒋᑦ? ᖃᐅᔨᒪᓕᕋᓗᐊᖅᐱᑎᒍᓪᓘᓐᓃᑦ? ᐃᒃᐱᖕᓂᐊᔪᖕᓃᓕᖅᐱᑕ?
- **ᐊᕙᓐ**: ᖁᔭᓐᓇᒦᒃ ᐅᖃᐅᓯᑐᔫᓐᓄᑦ ᑭᐊᕙᓐ. ᐃᒃᐸᒃᓴᖅ ᐊᔾᔨᕐᓚᖕᒋᓐᓂᒃ ᑐᓴᓚᐅᖅᑐᒍᑦ ᑲᖕᒋᖅᓚᓂᕐᒥᑦ ᐱᔾᔪᑎᖅᑲᖅᑐᓂᒃ ᐃᓄᐃᑦ ᖁᕕᐊᓱᙱᓐᓂᖕᒋᓐᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐋᖅᑭᐅᒪᔪᒥᑦ, ᐊᒥᓲᓂᖅᓴᐅᓕᖅᑐᓂᒃ ᓇᓄᕐᓂᑦ ᓄᓇᒥᑦ, ᐃᓄᐃᑦ ᑎᑭᐅᑎᖃᑦᑕᕐᓂᖕᒋᓐᓂᒃ ᓇᓄᕐᓄᑦ, ᐃᓄᐃᑦ ᓄᓇᓕᐊᕐᓂᖅ ᐊᔪᖅᑐᓂᒃ ᒪᖃᐃᕝᕕᒋᕙᓚᐅᖅᑕᖕᒋᓐᓄᙵᐅᓗᑎᒡᓗ ᐊᒻᒪᓗ ᖁᕕᐊᒋᓗᒍ ᐊᐅᔭᐅᔪᖅ ᓇᓄᐃᑦ ᐊᒥᓲᓗᐊᓕᕐᓂᖕᒋᑦ ᐱᔾᔪᑎᒋᑉᓗᒋᑦ ᐊᒻᒪᓗ ᓄᑕᖅᖃᖕᒋᓐᓂᑦ ᐊᓃᖁᔾᔭᙱᑦᑐᑦ ᐊᒻᒪᓗ ᐅᓗᕆᐊᓇᕐᓂᖅᓴᐅᓕᕐᓂᖕᒋᑦ. ᐅᖃᐅᓯᐅᖃᑦᑕᖅᑐᑦ ᐊᔾᔩᕐᓚᐅᔪᑦ ᐅᖃᑦᓗᕆᒃᓴᐅᑎᓂᕐᓂᑦ ᐃᓚᐅᖃᑕᐅᕝᕕᒋᓯᒪᔭᑉᑎᖕᓂᑦ ᐊᒻᒪᓗ ᑕᒪᓐᓇ ᑎᑎᕋᐅᓯᐅᖓᑦ ᓂᐱᓕᐅᖅᑕᐅᖓᑦ, ᑕᐃᒪᓐᓇ ᑲᔪᓯᓂᐊᖅᑐᖅ ᑎᑭᕋᕐᔪᐊᕐᒥᑦ ᐊᒻᒪᓗ ᐱᔭᐅᓂᐊᓗᓂ ᓄᓇᕗᒻᒥᑦ ᐆᒪᔪᓕᕆᔨᔪᐊᒃᑯᓐᓄᑦ ᐊᒻᒪᓗ ᐃᓱᒪᓕᐅᕆᖅᑲᑦᑕᖅᑐᓄᑦ ᓇᓄᓪᓚᒃᓴᐃᑦ ᒥᒃᓴᓄᑦ ᐊᒻᒪᓗ ᐃᓚᒋᔭᐅᓗᓂ ᐊᖏᓂᖅᓵᐅᔪᒧᑦ ᐅᖃᓪᓗᓇᒃᓴᐅᑎᓂᕐᓄᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᕐᓂᒃᐸᑦ.

- **ᐋᓕᒃᓴ**: ᓄᖅᑲᖅᑎᓐᓂᐊᕐᓗᒋᑦ ᐅᖃᐅᓯᒃᑲ, 1500 ᐅᖓᑖᓄᑦ ᒪᖃᐃᑎᓂᒃ ᓇᓄᖅᓯᐅᕈᒪᔪᓂᒃ. 10-12-ᖏᓐᓇᐃᑦ ᑐᓂᔭᐅᖃᑦᑕᖅᑐᑦ ᓇᓄᒍᑎᓂᒃ, ᐊᒧᖅᓴᐃᓂᒃᑯᑦ. ᐊᒧᔭᐅᔪᑦ ᒪᕿᒃᓂᒃ ᐅᑉᓘᖕᓂᑦ 48 ᐃᑲᕋᓂᑦ ᓇᓄᖅᓯᐅᖃᑦᑕᖅᓱᑎᒃ ᐊᒻᒪᓗ ᑕᒪᓐᓇ ᐊᖏᔩᔪᖅ ᓄᓇᓕᖕᒧᑦ. ᑕᐃᒪᑐᖅ ᐊᒥᓲᙳᑎᐊᖅᓯᒪᔪᓂᒃ ᐃᓂᐊᖅᐳᒍᑦ ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ, ᑭᖑᓪᓕᖅᐳᒥᑦ ᓈᓴᐅᓂᐅᓚᐅᖅᑐᖅ ᓈᒻᒪᓚᐅᖏᓐᓂᖓᓄᑦ (ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ 0-ᙳᓚᐅᖅᑐᖅ). ᒫᓐᓇ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖅ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐃᒡᓗᕈᖕᓂᖃᙱᑦᑐᑦ. ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖅ ᐅᖃᖅᑐᖅ ᓇᓄᐃᑦ ᐊᒥᓲᖏᓐᓂᖅᐸᓪᓚᐊᔪᑦ. ᐊᐅᔭᐅᑎᓪᓗᒍ, ᐊᒥᓲᓂᖅᓴᐅᖃᑦᑕᓕᖅᑐᓂᒃ ᓇᓄᕐᓂᑦ ᑕᑯᖃᑦᑕᓕᖅᑐᒍᑦ. ᓯᒡᔭᒃᑖᖓᓂᑦ ᐊᕝᕙᐊᑦ. ᐅᒃᐱᕆᓂᖅ ᐊᔪᕐᓇᖅᑐᖅ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖅ.

ᐃᓚᐅᖃᑕᐅᔪᑦ ᖁᔭᓐᓇᒦᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᒪᔫᓂᖃᑕᐅᓚᐅᕐᒪᑕ, ᐊᒻᒪᓗ ᐃᓱᒪᒋᔭᖏᑦ ᑐᓂᔭᖏᓐᓄᑦ.

20:49 ᑲᑎᒪᓂᖅ ᐃᓱᓕᑦᑐᖅ

2021-ᒥᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᔪᒥᑦ ᓇᓄᐃᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐅᓂᑉᑳ ᒥᒃᓵᓄᑦ ᐅᖃᖅᑲᐅᑎᖃᕐᓂᕐᒧᑦ ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᑎᑭᕋᕐᔪᐊᖅ

ᐃᑲᓂ: ᑎᑭᕋᕐᔪᐊᕐᒥᑦ Hᐋᓪᓚᒃᑯᑦ ᑲᑎᒪᕝᕕᖓᑦ
ᐅᑉᓗᖅ/ᓇᒧᖓᓕᑎᓪᓗᒍ: ᓄᕋᐃᑦ 9, 2024
ᐱᒋᐊᕐᒪᑦ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 19:20
ᐃᓱᓕᑦᑐᖅ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 20:31

ᐃᓚᐅᖃᑕᐅᔪᑦ

ᕈᐊᐳᑦ ᐃᓄᐊᐱᒃ (ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐃᒃᓯᕙᐅᑕᖅ)
ᐊᓕᓴ ᐳᕼᐊᑦ, ᓇᓄᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ
ᐊᕙᓐ ᕆᑦᓯᑦᓴᓐ, ᓇᓄᓂᒃ ᐆᒪᔪᓕᕆᔨ, ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ
ᔭᓇᑕᓐ ᐱᓐ, ᐊᖓᔪᒃᖠᖅ ᓂᕐᔪᑏᑦ ᒥᒃᓵᓄᑦ ᐅᖃᐅᔾᔨᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ
ᑲᐃᓪ ᕈᐃᑦᓯ, ᐊᖓᔪᒃᖠᖅ ᓂᕐᔪᑎᓄᑦ ᐊᒻᒪᓗ ᐊᑐᐊᒐᒧᑦ ᐅᖃᐅᔾᔨᔨ, ᓂᕐᔪᑎᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐆᒪᔪᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ
ᐋᓕᕗ ᓯᑉᑕᓐ-ᐅᐃᓕᐊᑦ, ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᓂᕐᔪᓯᕆᔨ ᑎᑭᕋᕐᔪᐊᖅ

ᒪᒥᐊᓐᓃᑦ: ᕿᐳᕆᐅᓪ ᓂᕐᓗᖕᒐᔪᖅ, ᐊᖓᔪᖅᖄᖅ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓂᑦ ᐊᒻᒪᓗ ᕈᐃᒪᓐ ᒎᓯ, ᐱᕝᕕᐅᖃᑦᑕᖅᑐᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᑭᒧᐊᒃᑎᑦᑎᔨ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓂᑦ

ᐱᔾᔪᑎᒋᔭᖓ ᑲᑎᒪᓂᐅᑉ:

ᑐᓂᓯᓗᑎᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓄᑕᐅᓛᓂᒃ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒥᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᖃᑉᓯᐅᑎᒋᓂᖏᓐᓄᑦ ᐅᖃᐅᓯᐅᓯᒪᓂᖓᑐᑦ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓄᑦ ᐅᓂᑉᑳᒥᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖏᓐᓂᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᖁᔭᖏᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓱᓇᑐᐃᓐᓇᐃᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐊᐱᖅᖁᑎᒋᔭᐅᔪᑦ ᑕᒪᐃᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᓄᑦ. ᑲᑎᑦᑎᓗᓂ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᑭᖑᓪᓕᕐᒥᑦ ᐱᓕᕆᐊᖑᓂᐊᖅᑐᑦ ᐊᒻᒪᓗ ᖃᓄᐃᓕᐅᕈᑕᐅᒋᐊᕐᓂᐊᖅᑐᑦ:

ᓇᐃᑦᑐᒥᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (4-6 ᐱᓇᓱᐊᕈᓯᕐᓂᑦ ᑲᑎᒪᓚᐅᖅᖄᖅᑎᓪᓗᒋᑦ):

- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᓪᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓄᑦ ᐊᐱᖅᖁᑕᐅᔪᑦ
- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᓇᓗᓇᐃᕐᓗᒍ ᓇᓚᐅᑦᑎᔪᖅ ᑎᑎᖅᖃᓯᒪᓂᖓᑕ ᑎᖕᒥᒃᑯᑦ ᓈᓴᐃᖅᑕᐅᔪᑦ ᐊᑎᖏᑦ
- ᖃᕆᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᑦᑎᕝᕕᒋᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐊᒻᒪᓗ ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐊᒥᐊᒃᑯᐃᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᖏᓐᓄᓕᕆᓂᖅ

ᐊᑯᓂᐅᓂᓴᒧᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (2-3 ᑕᖅᕿᑦ ᐃᓗᐊᓂᑦ ᑲᑎᒪᓚᐅᖅᖄᖅᑎᓪᓗᒋᑦ):

- ᑐᓂᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐊᐅᓚᓐᓂᖏᓐᓂᒃ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ

ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ:

ᐊᐱᖅᖁᑏᑦ ᑲᑎᒻᒪᔪᓄᑦ ᑎᑭᓯᑎᑕᐅᖅᖄᖅᑎᓪᓗᒋᑦ ᑭᑑᖕᒪᖕᑲᑕ:

ᑲᑎᒪᒋᐊᓚᐅᖅᑐᑦᖃᐅᔨᑎᑕᐅᑉᓗᑎᒃ ᑲᑎᒪᔪᓃᖅᑕᐅᔪᓂᑦ. ᐊᓕᓴ ᐳᕼᐊᑦ ᐅᖓᓯᒃᓯᒋᐊᖅᓯᔪᒪᓚᐅᖅᑐᖅ ᐅᖃᑎᖃᕐᓂᕐᒥᒃ ᐅᖃᑑᑎᒃᑯᖕᓖᓛᕐᑐᓂ ᐅᕙᑑᓐᓂᑦ ᖃᐅᐸᓗᖕᓖᑦ ᐊᒥᓲᓂᖅᓴᓂᒃ ᑲᑎᒪᔨᓂᒃ ᑲᑎᒪᔪᓃᖅᑕᐅᓂᐊᕐᓗᑎᒃ. ᕈᐊᐳᑦ ᐃᓄᐊᐱᒃ ᐅᖃᖅᑐᖅ ᑲᑎᒪᔨᓂᒃ ᖃᐅᔨᒋᐊᕐᓂᐊᖅᑐᖅ ᒪᓂᒪᔪᓐᓇᕐᒪᖔᑕ ᑲᑎᒪᖅᑕᐅᓂᐊᕐᓗᑎᒃ ᖃᐅᒃᐸᑦ, ᑭᓯᐊᓂ ᑲᑎᒪᓂᖅ ᑲᔪᓯᑎᓪᓗᒍ ᐅᑉᓗᒥ ᓈᓪᒪᐅᑎᓂᖅ ᐊᔪᕐᒪᑕ. ᐊᓕᓴ ᐳᕼᐊᑦ ᐊᔾᔨᓂᒃ ᑕᐅᑐᒃᑎᑦᑎᔪᖅ ᐊᓯᐊᓄᐊᖅᐸᑦᑕᐊᑉᓗᑎᒃ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᐃᔪᒃᑯᑦ ᐃᓅᔪᓐᓃᓕᐅᖅᓯᓂᖓᓄᑦ ᐆᒪᔪᓕᕆᔨ ᒫᒃ ᑎᒃ. ᑕᑯᑎᑦᑎᓕᖅᑐᓂ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᐃᓃᑦ ᐃᓚᐅᑉᓗᑎᒃ ᐸᕐᓇᐃᓂᖅᖃᖅᑎᓪᓗᒋᑦ ᑎᑭᕐᑐᒍ ᑲᑎᑐᒃᑖᖅᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᓱᒪᓕᐅᕆᓂᖅ ᓈᓪᒪᒋᔭᐅᓚᐅᓪᓚᑦ ᒥᓂᑦᑕᒧᑦ ᐊᒻᒪᓗ ᓇᓐᓄᒍᑏᑦ ᑐᓂᐅᖅᑲᐅᑕᐅᓂᕐᒥᑦ ᓄᓇᓕᖕᓄᑦ. ᐅᖃᕐᓗᓂ ᐱᔾᔪᑎᖕᒪᓂᑦ ᑲᑎᒪᑎᑦᑎᓂᐅᑉ ᑐᓂᓯᓗᑎᒃ ᓇᐃᓈᖅᓯᒪᔪᓂᒃ ᖃᓄᐃᖓᓂᐅᓱᓂᒃ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᐊᖕᒪᖕᓂᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᐃᓱᒪᒋᔭᖕᒪᖕᓂᒃ ᑐᓴᕐᓂᐊᕐᓗᑎᒃ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᑦ ᐊᔾᔨᑦ

- ᐅᖃᐅᓯᒃᓴᓂᒃ ᑐᓂᓯᔪᖃᓚᐅᖏᑦᑐᖅ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᔪᑦ/ᐱᓕᕆᓃᑦ

- **ᕈᐊᐳᑦ**: ᖃᖓᒃᑯᑦ ᓈᓴᐃᖅᑕᖅᐱᓯ, ᐃᓚᐅᖅᑕᖅᐸ ᑰᒡᔪᐊᑉ ᖃᓂᑕᖓ?

- **ᐊᓕᓴ:** ᐄ. ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᔪᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᐃᔭᕌᖓᐸᑕ, ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᔪᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᐃᖃᑕᐅᖃᑦᑕᕐᒥᔪᒍᑦ ᐊᑕᐅᑦᑎᒃᑯᑦ. ᖃᐅᔨᒪᔪᒍᑦ ᑭᒡᓕᓕᐅᖅᓯᒪᔪᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᕙᒃᑐᖅᓯᒪᔫᖓᑕ, ᓇᓄᐃᑦ ᑕᐅᑐᙵᒋᒻᒪᔾᔪᒃ ᑭᒡᓕᓕᐅᖅᓯᒪᓂᖏᓐᓄᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓂᖃᖃᑦᑕᕆᖕᓇᖅᑐᑦ.

- **ᕈᐊᐳᑦ:** ᓈᓴᐃᖃᑦᑕᖅᐱᓯ ᓄᓇᒥᑦ?
- **ᐊᓕᓴ:** ᐄ. ᖃᐅᔨᒋᐊᕋᐸᑕ (ᐅᖃᐅᑎᒃᑯᑦ ᓄᕙᒡᔪᐊᕐᓇᖅᑕᖃᖅᑎᓪᓗᒎᓚᐅᕐᒪᑦ), ᓈᓴᐃᓚᐅᖅᑐᒍᑦ ᓄᓇᒥᑦ ᐱᕐᑯᔭᐅᓂᐸᑎᓐᓄᑦ ᐅᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ. ᐃᓚᐅᑎᓐᓇᓱᖃᑦᑕᖅᑕᕗᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᖃᐅᔨᓴᐃᓂᐸᑎᖕᓄᑦ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓱᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᖃᓄᐃᖓᓂᐅᔪᑦ

- **ᕈᐊᐳᑦ:** ᑲᑎᒪᖃᑎᒋᓚᐸᑎᒍᑦ ᑭᕙᓪᓕᕐᒥᑦ ᐅᒪᔪᓕᕆᔪᕐᔪᐊᒃᑯᑦ ᑲᖏᖅᓯᓂᕐᒥᑦ, ᑕᑯᔭᒃᓴᖅᑕᖃᓚᐅᖅᑐᖅ ᐱᔾᔪᑎᖃᖅᑐᒥᒃ ᓈᓴᐃᓂᕐᒥᑦ. ᑕᑯᔭᒃᓴᐅᓚᐅᖅᑐᖅ ᐊᓯᐅᔨᓚᐅᖅᑐᒍᑦ 500 ᓇᓄᕐᓂᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐱᐸᓗᑎᒃ 500-ᓂᑦ. ᐊᐅᓚᒃᑲᒻᒪᑎᐊᓗᐃᑦ.
- **ᐊᓕᓴ:** ᐄ, ᐅᐱᒋᓚᐅᖏᑕᒃᓴᑎᔭᓯ ᑕᑯᒐᐸᓯᐅᒃ ᖃᓄᐃᖓᓂᐅᔪᑦ? (ᕈᐊᐳᑦ ᓂᐊᕐᑯᓂ ᐊᑐᖅᑰᒍ ᐊᖏᖅᑐᖅ ᐊᖏᕐᑲᑕᐅᐸᓗᓂ).
- **ᐋᓕᕗ:** ᐊᓚᐃᓇᐃᑦ ᑕᒪᕐᒥᒃ ᐊᑕᐅᑦᑎᒃᑯᑦ ᐱᓕᑎᐊᖑᕐᑲᑦᑕᕐᒪᑕ ᐊᒻᒪᓗ ᐊᑕᐅᑦᑎᒃᑯᑦ ᐊᐅᓚᑦᑑᔭᕐᑲᑦᑕᖅᑰᑎᒃ.

ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔾᔪᑦ ᐊᒻᒪᓗ ᓯᓚᐅᐸ ᐊᓯᔾᔨᖅᐸᒡᓕᐊᔪᓕᕆᔾᔪᑦ (ECCC) ᖃᐅᔨᓴᐃᓂᖓᑦ

- **ᐊᕙᓐ:** ᓄᓇᓕᒃ ᐱᔪᒪᒃᐸᑦ, ᐱᓕᕆᖃᑎᒋᔫᓐᓇᖅᑕᐸᓯ ᐋᓕᕗ ᖃᐅᔨᓴᕐᓗᒋᑦ ᓂᕿᖏᓐᓂᑦ ᐲᖅᑕᐅᔪᓂᑦ ᐆᒃᑐᕋᐅᑎᓂᒃ ᑲᑎᖃᑦᑕᖅᑕᖓᓂᑦ. ᖃᐅᔨᓇᓱᒡᓗᒋᓪᓗ ᖃᓄᐃᓕᐅᖃᑦᑕᕐᓂᖏᑦ ᐱᓗᐊᖅᑐᒥᑦ ᐃᓗᐃᓗᑕᐅᔪᑦ ᓇᓄᕐᓂᑦ. ᓇᓕᐊᖑᒪᖔᑕ, ᓇᒧᙵᐅᖃᑦᑕᕐᒪᖔᑕ, ᓇᑭᙶᕐᒪᖔᑕ.
- **ᐋᓕᕗ:** ᐄ, ᐅᖃᐅᓯᑲᒃᑲᓐᓂᕐᓂᐊᖅᐸᕗᑦ.

- **ᕈᐊᐳᑦ:** ᐅᖃᐅᓯᖃᕐᕕᖓᐸᓯ ᑲᓚᓕᖅᑐᖅᓱᑎ/ᑖᓪᓕᖅᑐᖅᓱᑎᑦ, ᓱᓇᒥᒃ ᐅᖃᐅᓯᖅᐱᑦ?
- **ᐊᕙᓐ:** ᒪᕈᐃᖓᓕᔪᖅ ᑐᕋᖓᓕᔪᖅ ᐱᓕᕆᐊᕆᖃᑦᑕᖅᑕᕗᑦ, ᐃᓚᖏᓐᓂᑦ ᑖᓪᓕᖅᑐᐃᖃᑦᑕᖅᑐᒍᑦ ᐊᒻᒪᓗ ᖁᖓᓯᕈᓯᖅᑐᐃᖃᑦᑕᖅᑐᒍᑦ. ᖃᐅᔨᓴᐃᖃᑦᑕᖅᑐᒍᑦ ᓱᕐᓗ ᐋᓐᓂᐊᕐᓂᕐᓂᑦ, ᓱᕐᓗ ᑎᖕᒥᐊᓂᑦ ᐋᓐᓂᐊᖑᖃᑦᑕᖅᑐᓂᑦ, ᑕᐃᒪᓐᓇᐃᑦᑐᓂᑦ, ᑕᒪᒃᑯᓂᙵᑦ ᐃᓄᖕᓄᑦ ᐃᓱᒫᕈᑎᖏᔭᐅᔪᓂᑦ. ᑲᑎᑎᕐᑲᑦᑕᕐᒥᔪᒍᑦ ᐊᓯᖏᓐᓂᒃ ᐆᒃᑐᕋᖅᑕᐅᓂᐊᖅᑐᓂᒃ ᖃᓄᐃᖏᑦᑎᐊᕐᓂᖏᓐᓄᑦ ᖃᐃᔭᐃᓗᑕ ᐊᒻᒪᓗ ᐱᑦᑕᐅᖏᑦᑐᖅᑲᕐᒪᖔᑦ. ᑐᑭᓯᐊᓗᒋᑦ ᐱᑦᑕᐅᖏᑦᑐᐃᑦ ᑕᑎᐅᕐᒥᑦ ᐊᒻᒪᓗ ᓂᕐᔪᑎᓄᑦ ᓂᑎᔭᐅᔪᑦ, ᐊᒻᒪᓗ ᓱᓇᓂᒃ ᐃᓄᐃᑦ ᓂᑎᖃᑦᑕᕐᒪᖔᑕ ᐊᒻᒪᓗ ᖃᓄᑎᒋᐅᖕᒪᖔᑕ ᓂᕐᔪᑎᓃᑦᑐᑦ ᐊᒻᒪᓗ ᐊᔾᔨᒌᙵᒋᑦᑐᓂᑦ ᐆᒪᔪᓂᑦ. ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔾᔪᑦ ᐱᔭᒃᓴᐅᑎᖃᕐᓚᒃᑐᑦ ᑐᑭᓯᐊᓂᐊᕐᓗᒋᑦ ᓱᓇᕋᓗᐃᑦ ᐊᕙᑎᑦᑐᑦ ᐊᒻᒪᓗ ᖃᓄᖅ ᐱᑎᑦᑎᖕᒪᖔᑕ ᐃᓄᐃᑦ ᐊᒻᒪᓗ ᓂᕐᔪᑏᑦ ᐋᓐᓂᐊᖃᖅᑕᐃᓕᒪᓂᖏᓐᓂᑦ. ᐊᒻᒪᓗ ᖃᐸᔨᐅᓂᖏᓐᓂᓕᕆᖃᑦᑕᖅᑐᒍᑦ ᐅᑭᐅᕐᓗᖏᓐᓂᑦ. ᑐᑭᓯᐊᓗᒋᑦ ᐊᒥᓱᑎᒋᓂᖏᑦ ᐱᕈᖅᐸᓪᓕᐊᓂᖏᑦ ᐊᒻᒪᓗ ᐊᒃᑐᖅᓯᓂᐅᔪᑦ ᐊᒻᒪᓗ ᐊᓯᐊᙳᖅᐸᓪᓕᐊᔪᓂᒃ ᑕᑎᐅᐸ ᓯᑯᐊᓂᑦ.

- **ᐋᓕᕗ:** ᓱᓇᒥᑦ ᖃᐅᔨᓴᐃᖃᑦᑕᖅᐸᑦ ᓇᓄᐃᑦ ᒥᖑᐊᖅᑕᐅᓕᖓᓚᑕ ᑖᓪᑕᐅᓕᖓᓚᑕ? ᑕᑯᒃᓴᐅᓂᖅᓴᐅᕙᑦ ᓂᕆᓇᓱᒃᓯᐅᖅᑎᓄᑦ?
- **ᐊᕙᓐ:** ᖃᐅᔨᒪᙱᑦᑐᖓ ᖃᓄᑐᐃᓐᓇᖅ ᖃᐅᔨᓴᐃᓂᖅᑕᖃᕐᒪᖔᑦ ᑕᒪᑐᒨᖓ. ᒥᖑᐊᑦ ᐃᒪᕐᒧᑦ ᓄᖑᑎᕈᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ ᐊᑯᓂᐊᓗᒃ ᐃᓕᓯᒪᕐᑲᑦᑕᙱᑦᑐᑦ.

- **ᐋᓕᕗ:** ᖃᐅᔨᓴᐃᓃᑦ ᐊᑯᐸᓕᖅᓯᒪᔪᖃᐃᓐᓇᖅᐸᑕ, ᑲᑎᑕᐅᓂᐊᖅᐸᑦ ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᐊᖏᓪᓕᒋᐊᖅᓯᒍᓂ ᑲᑎᒍᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ?
- **ᐊᓕᓴ:** ᐃᓱᒪᙵᒋᑦᑐᒍᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᒃ ᑖᐸᑯᐊ ᐊᑕᐅᓯᐅᑎᓪᓗᒋ ᐅᕝᕙᓘᓐᓃᑦ ᐊᓯᐊᙳᕐᓗᒋᑦ ᑭᒡᓕᒋᔭᐅᔪᑦ, ᑭᓯᐊᓂ ᐃᓱᒪᒋᖃᑦᑕᖅᑕᕗᑦ ᑲᑎᒍᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓄᑦ ᐱᕐᑯᔭᐅᔪᓂᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᑕᐅᑦᑎᒃᑯᑦ, ᖃᓄᐃᖓᓂᐅᔪᑦ ᑕᒪᕐᒥᑦ ᑕᑲᓗᒋᑦ. ᓈᓴᐅᑏᑦ/ᑎᑎᖅᖃᐃᑦ ᐊᐸᓯᒪᓗᒋᑦ ᑕᐅᑐᒃᑕᐅᒃᐸᑕ, ᐊᒥᓲᖕᓃᑎᐊᕐᑯᓇᔭᖅᑕᕗᑦ ᑲᑎᒍᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᒻᒪᓗ ᐊᒥᓯᙳᑎᐊᕐᑯᓗᒋᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐱᔾᔪᑎᒋᐸᓗᒋᑦ “ᐊᒥᓲᔪᖕᓃᖅᑐᑦ” ᐊᒻᒪᓗ “ᐊᒥᓯᙳᑎᐊᖅᑐᑦ”. ᑭᒥᐊᓂ ᐊᑐᖅᑰᒋᑦ ᑕᒪᕐᒥᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ, ᑲᑎᒍᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᓱᕐᑯᓯᖅᑕᐅᕐᑯᙵᒋᑕᕗᑦ ᑕᒪᐃᓐᓄᑦ.
- **ᐋᓕᕗ:** ᖃᐸᓰᕐᓗᑎᒃ ᓈᓴᐃᒃᐸᑕ ᐊᓯᐊᙳᖅᓯᓇᔭᖅᐸ ᑭᓪᓕᐅᔪᓂᑦ? ᐱᖓᓱᓂᒃ ᓈᓴᐃᓯᒪᓕᖅᑐᓯ, ᑭᓯᐊᓂ ᐅᖃᕈᐸᑕ ᒫᓐᓇᒥᑦ 15 ᐅᑭᐅᓂᑦ ᐊᒻᒪᓗ ᑐᑭᓯᐅᒪᔾᔪᐃᑦ ᓱᕐᑯᓯᐊᖏᐸᑕ ᑕᑯᒃᓴᐅᑎᓐᓂᐊᕐᓗᒋᑦ ᐊᑕᐅᓯᕐᒧᑦ ᑲᑎᕐᒪᔪᓄᑦ, ᐊᓯᐊᙳᕈᒪᓇᔭᖅᐱᓯᐅᒃ ᑭᓪᓕᐅᔪᑦ?
- **ᐊᓕᓴ:** ᑭᐅᒪᓇᒍᖕᓇᙱᑕᕋ ᐊᖏᔫᖕᒪᑦ ᐅᖃᐅᓯᓇᔭᕙᑦ. ᑭᓪᓕᒋᔭᐅᔪᑦ ᐋᖅᑭᒃᑕᐅᓚᐅᖅᑐᑦ ᐊᑐᖅᑰᒋᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᐅᓚᐅᖅᑐᑦ ᑕᐃᐸᓯᒪᓂᒥᑦ ᐊᒻᒪᓗ ᐊᒥᓲᓂᖅᓴᐅᑎᕐᑰᒋᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓂᐅᖓᓂᑦ.
- ᐅᖃᕐᑐᑲᒃᓴᐅᑎᓂᖅᑕᖃᒃᑲᓐᓂᓚᐅᖅᑐᖅ ᑕᒪᑐᒨᖓ ᐊᒻᒪᓗ ᖃᓄᖅ ᐊᓯᐊᙳᖅᑕᐅᓂᐊᕐᑐᑎᒃ ᑭᓪᓕᒋᔭᐅᔪᑦ ᐃᓱᒪᓕᐊᖑᓂᐊᕐᒪᖓᑦ ᑲᓇᑕᒦᖓᔪᒃᑯᑦ ᐊᒻᒪᓗ ᓄᓇᕗᐊᕐᒦᖓᔪᒃᑯᑦ.

- ᐊᓕᓴ ᐅᖃᖅᑐᖅ ᖃᓄᖅ ᑎᖕᒥᓲᒃᑯᑦ ᓈᓴᐃᓃᑦ ᓂᕈᐊᖅᑕᐅᖃᑦᑕᕐᒪᖔᑕ ᑎᑭᐅᑎᓂᖃᓗᐊᖕᒋᓐᓂᖅᓴᐅᖕᒪᑦ ᐊᑐᖅᑕᐅᓂᖕᖓ ᓈᓴᖅᑕᐅᑎᓪᓗᒋᑦ ᓇᓄᐃᑦ. ᐃᓱᒪᓕᐊᖑᓚᐅᖅᑐᖅ ᐃᓱᒪᒋᐸᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐱᔪᒪᔭᕐᒋᑦ. ᐊᓯᕐᒋᑦ ᐊᑐᖅᑕᐅᔪᑦ ᐅᖃᐅᓯᐅᔪᖕᓇᖅᑐᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᓯᕗᓂᒃᓴᒥᑦ ᓈᓴᐃᓂᐅᓂᐊᖅᑐᓄᑦ.

- **ᐋᓕᕗ:** ᖃᐅᔨᔪᒪᓇᕋᔭᖅᑐᖅ ᓂᕿᖕᒋᓐᓂᑦ ᐱᓂᐊᕐᓗᓂ ᐊᓯᖕᒋᓐᓂᑦ ᓱᕐᓗ ᓴᓪᓕᕐᒥᑦ. ᖃᐅᔨᔪᒪᔪᖕᖓ ᐊᒥᓲᑎᒋᓂᖕᒋᑦ ᐊᒃᑐᐊᓂᖃᖃᑦᑕᕐᒪᖕᖔᑕ ᐱᖕᖓᖕᓇᖅᐸᓯᖕᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ.
- **ᐊᓕᓴ:** ᐸᕐᓇᒃᑐᒍᑦ ᐱᓕᕆᐊᕆᓂᐊᕐᓗᒍ ᑕᓯᐅᔭᖅ ᐅᑭᐅᖑᔪᖅ ᐊᒻᒪᓗ ᑐᑭᓯᐅᒪᔾᔪᑎᒃᑲᓂᕐᓂᒃ ᐱᓂᐊᖅᑐᒍᑦ ᑕᐃᑲᙳᖓᑦ. ᐃᓚᓯᔪᒪᔪᒍᑦ ᓂᕿᖕᒋᓐᓂᑦ ᐱᖃᑦᑕᕐᓂᐊᕐᓗᑕ ᑕᒪᑐᒨᖕᖓ ᖃᐅᔨᓂᐊᕐᓗᒋᑦ ᐊᐅᓚᓂᖕᒋᑦ ᐊᑯᓂᖕᒋᓐᓂᑦ ᑭᒡᓕᒋᔭᐅᔪᓂᑦ.
- **ᐋᓕᕗ:** ᒥᖅᑯᖕᒋᓐᓂᒃ ᐲᔭᐃᓂᐊᖅᐱᓯ ᐱᖃᑦᑕᕐᒪᑕᑐᑦ ᐊᒃᑲᓄᑦ?
- **ᐊᓕᓴ:** ᐄ ᐸᕐᓇᐅᑎᒋᔭᕗᑦ ᑕᐃᒪᓐᓇᐃᓕᐅᕐᓂᐊᕐᓗᑕ ᐱᒋᐊᕐᓗᓂ ᐅᑭᐅᖑᔪᖅᑕᐅᖅ ᑕᓯᐅᔭᕐᒥᑦ.

- ᐊᓕᓴ ᑕᑯᑎᑦᑎᔪᖅ ᓴᓇᔭᖕᖓᓐᓂᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔾᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔾᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᑦᑐᖅᓴᕆᓪᔪᐊᖅ ᐃᐋᕗᑕᒥᑦ ᓇᓄᐃᑦ ᖁᖓᓯᕈᑎᖕᒋᓐᓂᑦ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᖃᓂᒃ.

- **ᐋᓕᕗ:** ᐊᖏᑎᒋᓂᖕᖓ ᓇᓅᑉ ᓇᓗᓇᐃᖅᓯᖃᑦᑕᖅᐸ ᓇᒧᙵᐅᖃᑦᑕᕐᒪᖕᖠᑦ?
- **ᐊᕙᓐ:** ᓂᕆᖃᑦᑕᖅᑕᖕᒋᓐᓂᐅᖃᑦᑕᖅᑐᖅ. ᓇᓄᐃᑦ ᐃᓂᐊᖑᖃᑦᑕᖅᑐᑦ ᓱᒃᖃᑦᑎᐊᖅᑐᒥᑦ ᓂᕆᑦᑎᐊᖃᑦᑕᖅᑐᑎᒃ ᐊᖏᒡᓕᓂᐊᕐᓗᑎᒃ.
- **ᐋᓕᕗ:** ᖁᖓᓯᕈᓯᖅᓯᖃᑦᑕᖅᐱᓯ ᐊᖑᓴᓪᓗᕐᓂᑦ?
- **ᐊᕙᓐ:** ᐋᒃᑲ. ᖁᖓᓯᓂᖕᒋᑦ ᐊᖑᓴᓪᓗᐃᑦ ᐊᖏᓗᐊᕐᒪᑕ ᖁᖓᓯᕈᑏᑦ ᐊᑕᓂᐊᕐᓗᑎᒃ. ᐊᕐᓇᓪᓗᐃᓐᓇᐅᔪᑦ ᐊᐅᓚᔪᒥᑦᑐᑦ. “ᒪᓂᙱᑦᑐᓂᒃ” ᐱᑕᖃᖅᑐᖅ ᒥᖅᑯᖕᒋᓐᓂᑦ, ᑭᓯᐊᓂ ᐊᔪᙱᓗᐊᕐᒋᑦᑐᑦ, ᓇᓄᕐᓄᑦ ᐱᓕᕆᐊᖑᒐᔮᒃᑐᑦ. ᓄᑖᓂᒃ ᓯᐅᑎᖕᒋᓐᓄᐊᖅᑕᐅᓲᓂᒃ ᐱᑕᖃᖅᑐᖅ ᐊᓐᑕᓇᖃᖅᖢᑎᒃ, ᑭᓯᐊᓂ ᐊᓐᑕᓇᐃᑦ ᓴᙶᑦᑐᑯᓗᐃᑦ ᓯᖁᒥᖃᑦᑕᖅᖢᑎᒃ. ᐊᖑᓴᓪᓗᕐᓄᐊᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᐃᓐᓇᕈᓕᖅᑐᓄᑦ. ᐊᑐᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᐱᕙᓪᓕᐊᓯᒪᔪᖅ ᐊᒻᒪᓗ ᐱᑦᑕᐅᓯᓂᖅᓴᐅᒃᐸᑕ ᕿᒥᕐᕈᔭᓇᕐᓂᐊᖅᑐᑦ ᐊᓯᖕᒋᓐᓂᑦ ᓃᐲᐊᕝᓄᑦ ᐊᑐᖅᑕᐅᔪᖕᓇᖅᑐᓂᒃ.

- **ᐋᓕᕗ:** ᐊᐅᓚᔪᖅ ᑐᓂᔪᖕᓇᖅᐱᓯᐅᒃ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ?
- **ᐊᓕᓴ:** ᐄ. ᐊᔾᔨᖕᖓᓂᑦ ᐊᐅᓪᓚᖅᑎᑦᑎᓂᐊᖅᐳᖕᖓ.
- **ᐊᕙᓐ:** ᐊᒥᓯᐃᑦ ᓇᓄᐃᑦ ᑲᑎᒋᐊᔭᑦᑐᑦ ᕿᑎᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ ᓯᑯᔭᕌᖓᑦ. ᑕᐃᒪᓐᓇ ᑕᑯᓯᒪᕖᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑕᐃᑯᙱᓚᐅᔪᖅᑲᖅᓯᒪᕙ? ᑕᐃᑯᙱᓚᐅᖅᓯᒪᙱᑦᑐᖕᖓ ᑕᐃᒪᓐᓇᒧᑦ ᖃᐅᔨᒪᙱᑦᑐᖕᖓ ᓱᓇᒧᑦ ᑕᐃᑯᙱᓚᐅᔾᔪᑎᑎᖃᖃᑦᑕᕐᒪᖕᖠᑕ.
- **ᕈᐊᐳᑦ:** ᖃᐅᔨᒪᙱᑦᑐᖕᖓ.
- **ᐊᕙᓐ:** ᐅᐱᓐᓇᖅᑐᖅ ᐊᔪᙱᓐᓂᖕᖓᑦ ᐃᖕᒋᖅᓴᔭᕐᓂᕐᒧᑦ, ᐊᖕᒋᔪᒪᕆᐊᓗᖕᒥᑦ ᐊᑐᖃᑦᑕᖅᑐᑦ. ᖁᖕᖓᓯᕈᓯᖅᓯᒪᖕᒪᑕ, ᑕᑯᔭᕗᑦ ᑖᑯᐊᑦᑕᐃᓐᓇᐃᑦ ᓇᓄᐃᑦ ᐊᑕᐅᓯᕐᒧᑦ ᐃᓂᒧᐊᖃᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᒥᑭᔪᓄᑦ ᕿᑭᖅᑕᓄᑦ ᐅᑭᐅᑕᒦᑦ.

ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᕐᒋᑦᑐᑎᒃ ᐱᓕᕆᐊᒃᓴᐃᑦ ᐱᖕᖓᖕᓇᖅᐸᓯᖕᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ

- ᐅᖃᓪᓗᓇᒃᓴᐅᑎᓚᐅᖅᑐᑦ ᓄᓇᕐᓂᑦ ᕿᒫᑎᑦᑎᔾᔪᑎᓂᒃ ᐊᒻᒪᓗ ᐱᓕᕆᐊᕐᓂᑦ ᑲᑎᑦᑎᓂᕐᒧᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᒃᑲᓂᕐᓂᑦ ᓂᕿᖕᒋᓐᓂᑦ ᐱᖃᑦᑕᕐᓂᒃᑯᑦ ᐊᒻᒪᓗ ᑕᐃᒪᐅᔪᓂᒃ ᓇᓄᐃᑦ ᓇᑭᙵᖅᓯᒪᓂᖕᒋᓐᓄᑦ ᖃᓇᑕᐅᔭᒃᑯᑦ ᑐᖅᑯᖅᓯᒪᔪᓂᑦ ᐱᖕᖓᖕᓇᖅᐸᓯᖕᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᕐᓄᑦ.
- ᐅᖃᓪᓗᓇᒃᓴᐅᑎᖃᓚᐅᖅᑐᑦ ᐆᒃᑐᑎ ᖃᐅᔾᔭᐃᓂᕐᒧᑦ ᐊᕿᐊᓂᑦ ᐆᒃᑐᕋᐃᔪᑦ ᕿᒫᑎᑦᑎᔾᔪᑎᓂᒃ ᐊᔾᔨᒌᙱᑦᑐᓂᑦ.
- **ᐋᓕᕗ:** ᕈᐊᐸ ᖃᕝᕙᐃᑦ ᐃᑲᔪᓗᐊᙱᑦᑐᑦ ᐅᕙᓪᓄᑦ. ᑐᖅᓱᓛᕈᑏᑦ ᐃᓛᓐᓂᒃᑯᑦ ᐊᔪᙱᖅᑲᑦᑕᖅᑐᑦ. ᐊᖏᔪᐃᑦ ᓇᓄᐃᑦ ᓲᖅᑯᑎᒃᓴᓗᐊᕐᒋᑦᑐᑦ, ᖃᐅᔨᓴᑐᐃᓐᓇᖅᑲᑦᑕᕋ ᑭᓯᐊᓂ ᐊᐅᓪᓚᖅᐸᑦ. ᐊᑐᖅᖢᒍ ᐅᑭᐅᕐᒋᑦ, ᑎᒥᖕᒋᑦᑕ ᖃᓄᐃᓐᓂᓇᔭᕐᒋᑦ. ᒥᑭᑦᑐᑦ ᓇᓄᐃᑦ ᖃᐅᔨᔪᒪᖃᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᒪᓕᒃᖢᑎᒃ ᐃᓄᖕᓂᑦ ᐃᒡᓗᖅᔪᐊᕐᒋᓐᓄᑦ, ᑭᓯᐊᓂ ᑲᐸᐊᔾᔮᑎᒃᑯᒃᑎᒃ, ᐊᔪᙱᑦᑐᖅᑕᐅᖅ. ᓴᓪᓘᔪᑦ ᓱᖁᑎᒃᓵᔭᕈᕐᓃᖅᑐᑦ. ᐊᔾᔨᒌᙱᑦᑑᑕᐅᔪᑦ ᑕᒪᒃᑯᓄᙵᓗᒃᑖᖅ.

ᑭᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓅᒃᑕᐅᔪᕐᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ ᐅᖃᓪᓗᓇᒃᓴᐅᑎᓃᑦ

- **ᕈᐊᐳᑦ:** ᐱᒃᑲᓂᕋᓯᓕᖅᑐᒧᑦ ᑖᒡᓂᑦ ᐊᑯᓂᕈᓗᒃ.
- **ᐋᓕᕗ:** ᐊᑐᙱᓚᐅᖅᑐᓄᑦ ᐱᖃᖅᑎᑕᐅᔪᓄᑦ. ᖃᓄᖅ ᐊᔪᙱᑉᐸ?
- **ᐊᓕᓴᐅᑉ** ᐅᖃᐅᓯᕆᔭᕐᒋᑦ ᑐᓂᔭᐅᖃᑦᑕᕐᓂᖅ HACCS-ᒥᑦ. ᐅᖃᐅᓯᕆᔭᕐᒋᑦ ᑐᓂᔭᐅᖃᑦᑕᖅᑐᑦ ᑲᑎᑉᐸᓪᓕᐊᓂᕐᒋᑦ ᐅᖃᒃᑲᓐᓂᖅᓗᓂᓗ ᑖᕙᓃᒃᑲᐸ ᐅᖃᐅᔾᔨᓂᐊᕐᓗᑕ.
- **ᐋᓕᕗ:** ᓱᓇᒥᒃ ᐱᓕᕆᐊᖃᖅᐸᑦ WWF-ᑯᑦ? ᐃᑲᔪᖅᐸᑦ ᐃᓕᑉᓯᕐᒋᓐᓂᑦ ᐅᕙᓘᓐᓃᑦ ᓱᓇᑐᐃᓐᓇᓂᑦ ᓈᓴᐅᑎᓄᑦ/ᑎᑎᖅᖃᓄᑦ ᑲᑎᑦᑎᖃᑦᑕᕐᒪᑕ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᖃᓂᒃ ᑖᕙᓂ, ᐊᑕᐅᑦᑎᒃᑯᑦ ᑐᕌᖓᔪᒥᒃ ᐱᖃᖅᑐᑦ. ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐱᓕᕆᖃᑎᖃᖅᑐᑦ WWF-ᑯᓐᓂᑦ ᓄᓇᕐᓂᑦ ᖃᐅᔨᓴᐃᔨᓄᑦ. ᐃᖕᒥᖕᓂᒃ ᐱᓕᕆᐊᖃᖃᑦᑕᖅᑐᑦ ᑕᑯᔭᐅᔪᓂᒃ ᐅᓂᑉᑳᓂᒃ ᐊᒻᒪᓗ ᐅᕝᕙᓄᐊᖃᑦᑕᖅᑕᖕᒋᑦ.
- **ᐊᕙᓐ:** ᐊᐱᖅᑯᓯᓇᓚᐅᖅᓯᒪᙱᑕᕗᑦ.

- **ᐋᓕᕙ**: ᖃᐅᔨᒋᐊᕐᕕᒋᓗᒍ ᔭᐃᓴᓐ H ᐊᕋᓯᒨ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᐱᔪᒪᒍᒃᑎᒃ. ᐱᑦᑕᐅᔪᖅ ᑐᕌᖓᔪᖅ, ᐱᓗᐊᖅᑐᒥᑦ ᐊᓯᖏᓐᓄᑦ ᓄᓇᓕᖕᓄᑦ ᐱᔭᐅᖃᑦᑕᖅᐸᑦ. ᓇᓄᕐᓂᒃ ᕿᒫᑎᖕᖓᑦ ᐅᕙᓂᒃ ᐃᑲᔪᖅᑐᖅ ᓯᓂᒍᖕᓇᕐᓂᐊᕐᓗᖕᖓ ᐅᓐᓄᒃᑯᑦ.

- **ᕈᐊᐳᕐ**: ᓇᓄᕐᒥᒃ ᕐᑯᖓᓯᕐᑯᑎᖃᖅᑐᒥᒃ ᓯᕐᑯᖅᑎᑕᐅᔪᖃᓚᐅᖅᑐᖅ, ᐱᓯᒪᕙᖓᑦ?
- **ᐊᕙᓐ**: ᓇᓄᖅ ᕐᑯᖓᓯᕈᓯᖅᑕᐅᓚᐅᖅᑐᖅ ᐊᒥᕋᐃᔭᕐᕕᒃ 2022-ᒥᑦ ᐊᒻᒪᓗ ᑕᑯᔭᐅᑉᓗᓂ ᓂᒋᐊᓂᑦ ᑰᔾᔪᐊᑉ ᐊᒥᕋᐃᔭᕐᕕᖕᒥᑦ ᑖᕙᓂ ᐅᑭᐅᕐᒥᑦ, ᐊᒻᒪᓗ ᐊᑕᐅᓯᕐᒥᑦ ᐅᑭᐅᓕᖕᒥᑦ ᐊᑎᖅᑕᕐᑲᓚᐅᖅᑐᖅ. ᐊᑎᖅᑕᖓ ᐱᑕᕐᑲᓚᐅᖕᒋᑦᑐᖅ ᓇᓐᓄᒃᑕᐅᖕᒪᑦ. 24-ᓂᒃ ᐅᑭᐅᕐᑲᓚᐅᖅᑐᖅ (ᑕᑯᑎᑕᐅᔪᖅ ᕐᑯᖓᓯᕈᑎ ᐃᓂᒋᔭᖕᖓᓂᑦ). ᐱᓱᓚᐅᖅᑐᖅ ᐋᒃᐸᑑᕐᔪᐊᕐᒧᑦ ᐊᒻᒪᓗ ᐅᑎᖅᓱᓂ.
- ᐋᓕᕙ ᐊᐱᕆᔪᖅ ᓄᓇᒻᒧᐊᖅ ᒪᕐᑲᐃᑎᒧᐊᖅᑕᐅᕐᑯᑉᓗᒍ. ᓯᕗᓪᓕᖅ ᓇᓄᖅ ᒪᕐᑲᐃᑎᒧᑦ. ᑐᓂᓯᒃᑲᓐᓂᖅᑐᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᒃᑲᓐᓂᕐᓂᑦ ᓇᓐᓄᒃᑕᐅᔪᓂᑦ. ᐃᓱᒪᔪᖅ ᐱᐅᓇᔭᕐᓂᖓᓄ ᑐᓂᔭᐅᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᒧᑦ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᒪᕐᑲᐃᑎᒧᑦ.
- **ᕈᐊᐳᕐ**: ᐊᓯᖏᓐᓂᑦ ᐊᐱᖅᑯᑎᕐᑲᓚᐅᖕᒋᑦᑐᑦ ᑭᓯᐊᓂ ᐊᓯᖏᑦ ᑲᑎᒪᔩᑦ ᑖᕝᕙᓃᕐᑯᓂᐊᖅᐸᒃᑲ ᖃᐅᒃᐸᑦ ᐊᒻᒪᓗ ᐊᔪᕐᓇᖅᐸᑦ ᖃᐅᔨᑎᓐᓂᐊᖅᐸᑎᑦ.

20:31 ᑲᑎᒪᓂᖅ ᐃᓱᓕᑦᑐᖅ

2021-ᒥᑦ ᐱᖓᖕᓇᖅᐸᓯᖕᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᔪᒥᑦ ᓇᓄᐃᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔾᓴᖅᑕᐅᓂᖕᒋᓐᓄᑦ ᐅᓂᑉᑳᑉ ᒥᒃᓵᓄᑦ ᐅᖃᖅᑎᖅᑲᕐᓂᕐᒧᑦ ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᑎᑭᕋᕐᔪᐊᖅ

ᐃᑲᓂ: ᑎᑭᕋᕐᔪᐊᕐᒥᑦ Hᐋᓪᓚᒃᑯᑦ ᑲᑎᒪᕝᕕᖓᑦ
ᐅᐸᑦᑐᖅ/ᓇᒧᖓᓕᑎᓪᓗᒍ: ᓄᕋᐃᑦ 10, 2024
ᐱᒋᐊᕐᒪᑦ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 10:40
ᐃᓱᓕᑦᑐᖅ ᖃᐅᔨᓴᐅᑎᒃᑯᑦ: 11:45

ᐃᓚᐅᖃᑕᐅᔪᑦ

ᒘᓇ ᐅᑲᓕᖅ (ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᑎᑎᕋᖅᑎ)
ᕈᐊᐳᑦ ᐃᓄᐊᐱᒃ (ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᐃᒃᓯᕙᐅᑕᖅ)
ᔩᓚ ᐃᐱᖕ
ᒪᓄ ᓇᑦᑕᑦ
ᐊᓕᓴ ᐳHᐊᑦ, ᓇᓄᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ
ᔭᓇᑕᓐ ᐱᓐ, ᐊᖓᔪᒃᖄᖅ ᓂᕐᔪᏗᑦ ᒥᒃᓵᓄᑦ ᐅᖃᐅᔾᔨᔨ, ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯ
ᑲᐃᓪ ᕈᐃᑦᓯ, ᐊᖓᔪᒃᖄᖅ ᓂᕐᔪᑎᓄᑦ ᐊᒻᒪᓗ ᐊᑐᐊᒐᒧᑦ ᐅᖃᐅᔾᔨᔨ, ᓂᕐᔪᑎᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐆᒪᔪᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ (NWMB)
ᐋᓕᕙ ᓯᐸᑕᓐ-ᐅᐃᓕᐊᑦ, ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᓂᕐᔪᓯᕆᔨ ᑎᑭᕋᕐᔪᐊᖅ

ᒪᒥᐊᓐᓃᑦ: ᕖᐳᕆᐅᓪ ᓂᕐᑐᖓᔪᖅ, ᐊᖓᔪᖅᑳᖅ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓂᑦ ᐊᒻᒪᓗ ᕈᐃᒪᓐ ᒘᓯ, ᐱᕝᕕᐅᖃᑦᑕᖅᑐᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᑭᒧᐊᒃᑎᑦᑎᔨ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓂᑦ

ᐱᔾᔪᑎᒋᔭᖓ ᑲᑎᒪᓂᐅᑉ:

ᑐᓂᓯᓗᑎᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓄᑕᐅᓛᓂᒃ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᕐᒥᑦ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐱᖓᖕᓇᖅᐸᓯᖕᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᓇᓄᐃᑦ ᖄᓯᐅᑎᒋᓂᖕᒋᓐᓄᑦ ᐅᖃᐅᓯᐅᓯᒪᓂᖕᖓᑦ 2021-ᒥᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖕᒋᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓄᑦ ᐅᓂᑉᑳᒥᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓇᓱᒃᑐᓕᕆᓂᖕᒋᓐᓂᑦ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᑎᓗᒃᑖᖅᓱᒥᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᖕᒋᓐᓄᑦ ᐊᑐᖅᑕᐅᖁᔭᕐᒥᑦ. ᐅᖃᐅᓯᕆᓗᒋᑦ ᓱᓇᑐᐃᓐᓇᐃᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᐊᐱᖅᑯᑎᒋᔭᐅᔪᑦ ᑕᒪᐃᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᓄᑦ. ᑲᑎᑦᑎᓗᓂ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖕᒋᑦ ᐱᖓᖕᓇᖅᐸᓯᖕᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᑭᖑᓪᓕᕐᒥᑦ ᐱᓕᕆᐊᖑᓂᐊᖅᑐᑦ ᐊᒻᒪᓗ ᖃᓄᐃᓕᐅᕈᑕᐅᒋᐊᕐᓂᐊᖅᑐᑦ:

ᓇᐃᑦᑐᒥᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (4-6 ᐱᓇᓱᐊᕈᓯᕐᓂᑦ ᑲᑎᒪᓚᐅᖅᖄᖅᑎᓪᓗᒋᑦ):

- ᖃᓇᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᓪᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᓐᓄᑦ ᐊᐱᖅᖁᑕᐅᔪᑦ
- ᖃᓇᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᓇᓗᓇᐃᕐᓗᒍ ᓇᓚᐅᑦᑎᔪᖅ ᑎᑎᕋᖅᓯᒪᓂᖓᑕ ᑎᖕᒥᓲᒃᑯᑦ ᓈᓴᐃᖅᑕᐅᔪᑦ ᐊᑎᖏᑦ
- ᖃᓇᑕᐅᔭᒃᑯᑦ ᑎᑎᖅᖃᑎᒍᑦ ᐊᐅᓪᓚᖅᑎᑦᑎᕝᕕᒋᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐊᒻᒪᓗ ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᐊᒥᐊᒃᑯᐃᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᖏᓐᓄᓕᕆᓂᖅ

ᐊᑯᓂᐅᓂᖓᒍᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᑦ (2-3 ᑕᖅᕿᑦ ᐃᓗᐊᓂᑦ ᑲᑎᒪᓚᐅᖅᖄᖅᑎᓪᓗᒋᑦ):

- ᑐᓂᓗᒋᑦ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐊᐅᓚᓐᓂᖅᒋᓐᓂᒃ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ

ᑲᑎᒪᓂᕐᒥᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ:

ᐊᐱᖅᖁᑏᑦ ᑲᑎᒻᒪᔪᓄᑦ ᑎᑭᓯᑎᑕᐅᖅᖄᖅᑎᓪᓗᒋᑦ ᑭᓲᒻᒪᖔᑕ:

ᑲᑎᒪᒃᑲᓐᓂᓚᐅᖅᑐᑦ ᐊᒥᓲᓂᖅᓴᓂᒃ ᑲᑎᒪᔨᓂᒃ ᑲᑎᒪᖃᑕᐅᔪᖅᖄᓂᐊᕐᓗᓂ. ᑲᑎᒪᒋᐊᓚᐅᖅᑐᑦᖃᐅᔨᑎᑕᐅᑉᓗᑎᒃ ᑲᑎᒪᔫᖃᑕᐅᔪᓂᑦ. ᐊᓕᓴ ᐳᕼᐊᑦ ᐊᔾᔨᓂᒃ ᑕᐅᑐᒃᑎᑦᑎᔪᖅ ᐊᓯᐊᓄᐊᖅᐸᓪᓕᐊᑉᓗᑎᒃ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᐃᔨᒃᑯᑦ ᐃᓅᔪᓐᓃᓕᐅᕐᓂᖓᓄᑦ ᐆᒪᔪᓕᕆᔨ ᒫᑲᔅ ᑎᒃ. ᑕᑯᑎᑦᑎᓕᖅᖢᓂ ᐱᔾᔪᑎᖃᖅᑐᓂᒃ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᐃᓃᑦ ᐃᓚᐅᑉᓗᑎᒃ ᐸᕐᓇᐃᓂᖃᖅᑎᓪᓗᒋᑦ ᑎᑭᔅᖢᒍ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᓱᒪᓕᐅᕆᓂᖅ ᓈᒻᒪᒋᔭᐅᓚᐅᕐᒪᑦ ᒥᓂᔅᑕᒧᑦ ᐊᒻᒪᓗ ᓇᓐᓄᒍᑏᑦ ᑐᓂᐅᖅᖃᐅᑕᐅᓂᖏᑦ ᓄᓇᓕᖕᓄᑦ. ᐅᖃᖅᖢᓂ ᐱᔾᔪᑎᖓᓂᑦ ᑲᑎᒪᑎᑦᑎᓂᐅᑉ ᑐᓂᓯᓗᑎᒃ ᓇᐃᓈᖅᓯᒪᔪᓂᒃ ᖃᓄᐃᖓᓂᐅᔪᓂᒃ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᒻᒪᓗ ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔩᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔩᒃᑯᑦ ᐱᓕᕆᐊᖑᓐᓂᑦ ᐃᓱᒪᒋᔭᖑᓐᓂᒃ ᑐᓴᕐᓂᐊᕐᓗᑎᒃ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐱᔾᔪᑎᖃᖅᑐᑦ ᐊᔾᔩᑦ

- ᐅᖃᐅᓯᒃᓴᓂᒃ ᑐᓂᓯᔪᖃᓚᐅᖏᑦᑐᖅ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᔪᑦ/ᐱᓕᕆᓃᑦ

- ᐅᖃᐅᓯᒃᓴᓂᒃ ᑐᓂᓯᔪᖃᓚᐅᖏᑦᑐᖅ.

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᑎᒋᓂᖏᓐᓂᒃ ᓈᓴᖅᑕᐅᔪᓂᑦ ᖃᓄᐃᖓᓂᐅᔪᑦ

- **ᕿᐊᐳᕐ:** ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᓈᓴᐅᑏᑦ ᐱᒍᑉᓯᐅᒃ, ᑲᑎᓗᒃᑖᖅᓯᒪᔪᑦ ᐱᓯᒪᔭᓯ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑲᑎᒪᕙᑦ? ᖃᑉᓯᐅᑎᒋᓂᖏᑦ ᐊᐅᓚᑕᐅᓂᐊᕐᓗᑎᒃ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ?
- **ᐊᓕᓴ:** ᑕᑯᔭᑉᑎᒍᑦ ᒫᓐᓇ ᐊᐅᓚᓐᓂᐊᕐᓗᒋᑦ ᐊᑕᐅᓯᐅᓗᒋᑦ/ᐊᑐᓂ ᐱᖃᑦᑕᕐᓗᒋᑦ, ᑭᓯᐊᓂ ᒪᕐᕉᖕᒪᑎᒃ ᐊᑉᓯᒪᔫᒃ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ. ᖃᑉᓯᐅᓂᖏᑦ ᑕᑯᒍᑉᑎᒍᑦ, ᐊᓯᐊᙳᙱᑦᑐᖅ ᑭᖑᓪᓕᖅᐹᒥᑦ ᖃᐅᔨᓴᐃᓐᒪᑕ ᓈᓴᐃᓐᒪᑕ. ᐅᕙᑎᓐᓄᑦ ᐊᒥᓲᑎᒋᓂᖏᑦ ᐊᓯᐊᙳᙱᑦᑐᑦ ᐊᒻᒪᓗ ᓯᖁᓯᐊᔪᒥᓇᖏᕐᖢᑎᒃ. ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ.

- **ᕿᐊᐳᕐ:** ᖃᑉᓯᐅᓂᖏᑦ ᑕᑯᑉᓗᒋᑦ ᐊᒥᓲᓕᕐᓂᖅᐹᓕᖅᑑᔭᖅᑐᑦ.
- **ᐊᓕᓴ:** ᐊᖏᖅᑐᖅ. 2021 ᐊᓄᕌᖃᑦᑕᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᑎᖕᒥᓲᒃᑯᑦ ᓈᓴᐃᓃᑦ ᑭᒡᓕᖅᑲᑦᑕᓚᐅᖅᖢᑎᒃ.
- ᑕᑯᒃᓴᐅᑎᑕᐅᔪᑦ ᐱᔾᔪᑕᐅᔪᑦ ᓈᓴᖅᑕᐅᓂᐊᕐᓗᑎᒃ ᑕᒪᐃᓐᓄᑦ ᓄᓇᕐᓄᑦ ᐊᑕᐅᑦᑎᒃᑯᑦ.
- **ᐋᓕᕗ:** ᐃᓯᐊᓂᑦ ᐅᑉᐴᑉ, ᓇᓄᐃᑦ ᖃᓄᐃᖅᑎᑦᑐᑦ.
- **ᐊᓕᓴ:** ᐄ, ᐊᒻᒪᓗ ᐃᓯᒪᔪᖓ ᐊᕙᓐ ᖃᐅᔨᓴᐃᓂᖓ ᑕᑯᒃᓴᐅᑎᑦᑎᑦᑎᐊᖅᑐᖅ ᑕᒪᑐᒥᖓᑦ.

ᑲᓇᑕᒥᑦ ᐊᕙᑎᓕᕆᔩᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔩᒃᑯᑦ (ECCC) ᖃᐅᔨᓴᐃᓂᖓᑦ

- ᐊᐅᓚᔪᒥᒃ ᑕᑯᑎᑕᐅᑎᓪᓗᒥ, ᐊᓕᓴᐅᑉ ᐃᖅᖃᐃᑎᑕᕐᒥᑦ ᑭᑐᒃᑖᑦ ᖁᖓᓯᕈᑎᓂᑦ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᑕᑯᒃᓴᐅᔪᑦ ᐊᕐᓇᓪᓗᐃᓐᓇᕐᓂᑦ.
- **ᕿᐊᐳᕐ:** ᖃᖓᒃᑯᑦ ᐅᑭᐅᑉ ᐃᓗᐊᓂᑦ ᓄᓕᐊᖃᑦᑕᖅᐸᑦ? ᓇᑦᑎᐊᑦ/ᐄᐳᕆ?
- **ᐊᕙᓐ:** ᐄᐳᕆ/ᓄᕐᕋᐃᑦ

ᑭᑐᑐᐃᓐᓇᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᓐᓂᐊᖏᓪᓗᑎᒃ ᐱᓕᕆᐊᒃᓴᐃᑦ ᐱᖓᓐᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ

- ᐅᖃᐅᓯᒃᓴᓂᒃ ᑐᓂᓯᔪᖃᓚᐅᖏᑦᑐᖅ.

ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ ᒥᒃᓵᓄᑦ ᐅᖃᓪᓗᓇᒃᓴᐅᑎᓃᑦ

- ᐊᓕᓴ ᐅᖃᒃᑲᓐᓂᖅᑐᖅ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᓂᑦ ᐊᓯᐊᙳᖅᓯᖁᔨᙱᑦᑐᑦ
- **ᕈᐊᐳᑦ:** ᐊᒥᓱᙳᕆᐊᖅᑕᐅᖁᑉᓗᒋᑦ ᐊᐱᕆᓇᓱᖅᑲᑦᑕᖅᑐᒍᑦ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ. ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ, ᓱᓇᐅᕙ ᑲᑎᓗᒃᑖᖅᖢᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ?
- **ᐊᓕᓴ:** 25 ᓴᓂᑭᓗᐊᕐᒧᑦ. ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᐊᑐᖅᑐᑦ ᐋᓐᑎᐅᕆᔪᒧᑦ ᐊᒻᒪᓗ ᑯᐸᐃᒃᒧᑦ. ᓇᓄᖅᑐᖃᓗᐊᕈᖕᓇᙱᑦᑐᖅ ᐋᓐᑎᐅᕆᔪᒥᑦ ᑭᓯᐊᓂ TAT-ᖃᖅᑐᑦ 23-ᓂᒃ ᑯᐸᐃᒃᒧᑦ. ᑕᐃᒪᓕ 48-ᙳᓇᔭᖅᑐᑦ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ ᑲᑎᓪᓗᒋᑦ.

- **ᕈᐊᐳᑦ:** ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐃᓪᓗᓕᒑᕐᔪᖕᒧᐊᕌᖓᑕ, ᑖᒃᑖᖅᑎᑕᐅᙱᓐᓇᖃᑦᑕᖅᑐᑦ ᐃᓪᓗᓕᒑᕐᔪᒃ. ᓱᖕᒪᑦ ᐃᓪᓗᓕᒑᕐᔪᒃ ᑕᓯᐅᔭᕐᒦᑦᑐᖅ, ᓲᖅ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑖᒃᑖᖃᑦᑕᖅᐸᑦ?
- **ᐊᓕᓴ:** ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑭᓯᒥ ᐃᓱᒪᓕᐅᖃᑦᑕᖅᑐᑦ ᑭᑐᑦ ᑖᒃᑖᕐᓂᐊᕐᒪᖔ. ᖃᑉᓯᐊᔪᐃᑦ ᐅᑭᐅᑦ ᐊᓂᒍᖅᑐᓂᑦ ᖃᐅᔨᓚᐅᖅᑐᖓ ᖃᒪᓂᑦᑐᐊᖅ ᐊᒻᒪᓗ ᐃᓪᓗᓕᒑᕐᔪᒃ ᑕᓯᐅᔭᖅ ᑖᒃᙱᓐᓃᓐᓇᖅ ᐱᖃᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᑐᓴᕝᕕᓕᑕᐅᖅᓯᒪᙱᑦᑕᕗᑦ ᑭᕙᓪᓕᕐᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐊᓯᖏᓐᓄᐊᖅᓯᔪᓂᒃ. ᑲᑎᒪᖃᑎᖃᖅᑐᒍᑦ ᑭᓯᒥ ᓄᓇᓕᖕᓂᑦ ᒫᓐᓇ ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥᑦ ᑐᓂᔭᐅᖃᑦᑕᖅᑐᓂᑦ.

- ᑕᑯᑎᑦᑎᔪᖅ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᒥᑦ ᓇᓄᕐᒥᑦ ᓇᓐᓄᒃᑕᕆᔭᐅᓵᓚᐅᖅᑐᒥᑦ ᑎᑭᕋᕐᔪᐊᕐᒥᐅᑕᕐᒧᑦ.
- **ᓲᓚ:** ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᔪᖅ ᖁᖓᓯᕈᓯᖅᓯᒪᓚᐅᖅᐸ?
- **ᐊᕙᓐ:** ᒪᕐᕿᒃ ᐅᑭᐅᓄᑦ.
- **ᓲᓚ:** ᖃᐅᔨᒋᐊᖃᑦᑕᖅᐱᓯᐅᒃ ᓇᓄᐃᑦ ᖁᖓᓯᕈᓯᖅᓯᒪᑎᓪᓗᒋᑦ?
- **ᐊᕙᓐ:** ᖁᖓᓯᕈᑎ ᐲᖅᑐᒃᓴᐅᑕᐅᓚᐅᖅᑐᖅ ᐊᒥᕋᐃᔭᕐᕙᖕᒥᑦ. ᓴᑉᑯᐃᔪᖕᓇᖅᑎᑦᑎᔪᒥᒃ ᐱᑕᖃᖅᑐᖅ ᖃᖓᒃᑰᖓᔪᖃᖅᖢᓂ ᐲᕐᓂᐊᕐᓂᖓᓄᑦ. ᐃᓚᐅᓚᐅᙱᑦᑐᖓ ᖁᖓᓯᕈᑎᓕᖅᑕᐅᖕᒪ, ᑭᓯᐊᓂ ᐃᖁᖓᓯᕈᓯᖅᓯᓚᐅᖅᑐᖅ ᓱᒃᖓᑎᓗᐊᓚᐅᖅᑕᖓ, ᖃᐅᔨᒋᐊᕐᓂᐊᖅᑐᒍᑦ ᑕᐃᒪᓐᓇᐃᒃᑲᓐᓂᖅᑐᖅᑳᙱᓪᓗᓂ. ᐊᔾᔨᓂᒃ ᑕᑯᔭᒪᙱᑦᑐᒍᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑐᓴᖅᑐᑕ ᖁᖓᓯᕈᑏᑦ ᐊᒃᑐᖅᓯᓂᖅᓯᓂᙱᓐᓂᒃ ᓇᓄᕐᓄᑦ ᐊᒻᒪᓗ ᖁᔭᓐᓇᒦᒃ ᑐᓂᓯᕙᑦ ᑕᒪᑐᒧᖓ ᑐᑭᓯᐅᒪᔾᔪᑎᒥᒃ.
- **ᐊᕙᓐ** ᑕᑯᑎᑦᑎᔪᖅ ᐊᔾᔨᒥᒃ ᖁᖓᓯᕈᑎᒥᑦ ᐊᑐᖅᑕᐅᔪᒥᑦ.
- **ᓲᓚ:** ᐅᑉᓗᖃᕈᕝᕕᑦ ᖁᖓᓯᕈᑎ ᐲᓛᕐᓂᖓᓄᑦ, ᓇᓄᕐᒧᖓᐅᖃᑦᑕᖅᐱᓯ ᐲᒻᒪᓇᒃᑲᑐᐊᕐᒪᖔᑦ ᖃᐅᔨᓂᐊᕐᓗᒍ?
- **ᐊᕙᓐ:** ᐊᑐᖅᓱᒍ ᓇᓄᐃᑦ ᓇᓃᒻᒪᖔᑕ, ᐱᓕᕆᒍᑉᑕ ᖃᓂᑑᓂᑦ ᖃᐅᔨᒋᐊᖃᑦᑕᖅᑐᒍᑦ. ᖁᖓᓯᕈᑏᑦ ᓇᑐᓇᐃᖅᓯᔾᔪᑎᖃᖅᑐᑦ ᐊᒻᒪᓗ ᑐᓵᖃᑦᑕᖅᑕᕗᑦ ᐊᒻᒪᓗ ᖃᐅᔨᒋᐊᖅᓱᒍ ᖃᓂᒻᒪᖓᑕ. ᖁᖓᓯᕈᑎ ᐋᖅᕿᒃᑕᐅᓯᒪᒪᓚᑐᐊᖅᑎᓪᓗᒍ ᐲᕐᓂᐊᕐᓗᓂ, ᑐᓴᓇᓯᖃᑦᑕᖅᑕᕗᑦ ᓱᓕ ᑭᖑᓂᐊᒍᑦ ᐅᑭᐅᓂᑦ ᐲᒻᒪᓇᒃᑲᑐᐊᕐᒪᖔ. ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓂᑦ ᐊᔪᖅᓇᖅᑎᑕᐅᔾᔪᑎᒥᑦ ᐳᒥᑦᒥᑦ ᐱᖃᙱᓐᓇᐸᑕ ᐱᓕᕆᓂᐊᖅᑐᑕ ᑕᒃᐸᓂ ᑭᓯᐊᓂ ᐱᑦᑎᐊᙳᓇᕐᓯᖃᑦᑕᖅᑐᒍᑦ.
- **ᓲᓚ:** ᓱᖃᑦᑕᖅᐱᓯᐅᒃ ᖁᖓᓯᕈᑎ?
- **ᐊᕙᓐ:** ᐊᐅᓪᓚᖅᑎᖃᑦᑕᖅᑕᕗᑦ ᐊᑐᖅᑕᐅᒃᑲᓐᓂᕈᖕᓇᖅᓯᑎᕐᓱᒍ. ᖃᕆᑕᐅᔭᒃᑯᑦ ᓈᓴᐅᑎᓂᒃ/ᑎᑎᖅᑲᓂᒃ ᐱᖃᑦᑕᕐᒥᔪᒍᑦ ᖁᖓᓯᕈᑎᒥᖔᖅᑐᒥᑦ. ᖃᓕᑦᑕᖅᑎᑕᐅᓯᒪᔪᒥᑦ, ᑭᓯᐊᓂ ᑐᑭᓯᐅᒪᔾᔪᑎᓗᒃᑖᑦ ᑐᓂᔭᐅᖃᑦᑕᖅᑐᑦ (ᐊᑐᖅᑕᐅᔪᖅ ᑭᒡᓕᖅᑲᖅᑐᖅ). ᓈᓴᐅᑏᑦ/ᑎᑎᖅᑲᐃᑦ ᑐᖅᑯᒃᑕᐅᖃᑦᑕᖅᑐᑦ ᐃᖅᑲᐅᒪᔪᕐᓇᖅᑐᒧᑦ ᖁᖓᓯᕈᑎᒥᑦ ᐃᒧᑦᑕᖅᑐᖅ ᖃᕆᑕᐅᔭᒃᑯᑦ ᐱᔪᕐᓇᖅᓱᑎᒍᑦ.
- **ᕈᐊᐳᑦ:** ᐊᓕᐊᓇᐃᖅᑕᖅᑐᖅ ᑕᑯᑎᑦᑎᔪᖕᓇᖅᓯᓂ ᐊᐅᓚᓐᓂᓇᔭᖅᑐᓂᑦ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᓂᑦ ᓇᓄᐃᑦ. ᐊᔾᔨᓂᒃ ᑕᑯᓪᓚᕆᙳᓚᐸᑕ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᓂᒃ ᑕᐃᒪᓐᓇᑐᑦ, ᐱᐅᒋᙱᑕᕋ. ᑐᑦᑐᑦ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᓂᒃ ᑐᑦᑐᓂᒃ. ᐊᓯᖏᓐᓂᒃ ᐃᓄᓐᓄᑦ ᑐᓴᖅᑕᐅᔪᓂᒃ ᐱᔪᓐᓇᖅᑐᑦ ᓄᓇᓕᖕᓂᑦ ᖁᖓᓯᕈᓯᖅᑐᐃᓂᐊᕐᓗᓂ. ᐅᖃᐃᓐᓇᕐᑕᖃᑦᑕᖅᑐᒍᑦ ᐱᐅᒋᙱᑕᕗᑦ ᑕᐃᒪᓐᓇ, ᑭᓯᐊᓂ ᖃᐅᔨᒪᔪᒍᑦ ᑕᐃᒪᓐᓇᐃᑕᐅᕐᓯᓂᐊᕐᒪᑕ. ᖃᓄᐃᓕᖓᓂᖓᓐᓂᒃ ᐅᖃᐅᓯᖃᓲᖅᑐᒍᑦ ᑕᒪᓐᓇ, ᐃᓯᒪᔪᖅ. ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑕᐃᒪᓐᓇᐃᑕᐱᓚᒃᐸᑕ, ᖃᓄᐃᑦᑐᐊᕐᓂᖅ ᐊᔪᖅᑐᒍᑦ.
- **ᐊᕙᓐ:** ᐃᓂᒥᑦ ᐊᐅᓚᑦᑎᓂᖅᓴᖅᑐᓂ ᐱᔾᔪᑎᖃᖅᑐᓂ ᐃᓚᖅᑭᖅᓯᓂᕐᒧᑦ ᐱᑕᖃᖅᑐᓂ. ᖁᖓᓯᕈᓯᖅᑐᐃᓂ ᐱᖃᑦᑕᖅᑐᒍᑦ ᒫᓂᑐᐊᖅ, ᒫᓂᑐᐊᖅ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐱᔪᓐᓇᖅᑎᑦᑎᔾᔪᑎᒋᔭᖅᑐᑦ ᐳᒥᓂᑦ. ᖃᐅᔨᒪᔪᖓ ᐱᓕᕆᐊᖑᒻᒪᑦ ᐃᓄᖕᓂᑦ ᑭᒡᓕᖅᑐᐃᓂᐊᖅᑐᓂ ᑲᑎᒪᔨᓄᑦ ᐃᓂᖕᓂᑦ ᒫᓂᑐᐊᖅ. ᖁᖓᓯᕈᓯᖅᑐᐃᓂᐊᕐᓂᖅᓴᐅᓂᖓᓐᓂᒃ (ᐊᒥᓱᓕᕐᓗᒋᑦ ᖁᖓᓯᕈᓯᖅᓯᒪᔪᑦ) ᐊᒻᒪᓗ ᐊᖏᓗᐊᖏᑦᑐᓂᒃ ᑐᑭᓯᐅᒪᔾᔪᑎᓂᒃ ᐱᓂᐊᕐᓗᑕ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐱᔭᕇᖅᑐᓂ. ᓈᒻᒪᒋᐊᖃᖅᑐᓂᒃ ᐃᓗᓕᕐᓂᐊᕐᓗᒍ ᐱᔭᕇᖅᑐᓂ ᓄᓇᓕᖕᓂ ᐃᓕᓴᐃᒍᑎᒋᔪᓐᓇᖅᑕᕗᑦ, ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓴᕐᓂᓕᕆᓂᕐᒧᑦ ᐱᔭᕇᖅᑐᓂᒃ. ᐃᓗᓕᕐᓂᕐᒥᒃ ᐱᓇᓱᖃᑦᑕᖅᑐᒍᑦ. ᐱᑎᑦᑎᓗᒋᑦ ᐊᑐᖅᓯᖃᑦᑕᖅᑐᒍᑦ.
- **ᕈᐊᐳᑦ:** ᖃᐅᓯᓂᒃ ᐅᑭᐅᓂᒃ ᖁᖓᓯᕈᓯᖅᑕᐅᖃᑦᑕᖅᐸ?
- **ᐊᕙᓐ:** ᐅᑭᐅᑕᒫᑦ ᖁᓕᓂᒃ ᖁᖓᓯᕈᓯᖅᓯᖃᑦᑕᖅᑐᒍᑦ. ᐱᓕᕆᐊᖑᓯᒪᔪᖅ ᐊᓯᖏᓐᓂᒃ ᐃᓕᓯᖃᑦᑕᖅᑐᒍᑦ. ᐃᓚᒋᔭᐅᔪᖅ ᑲᔪᓯᑎᑦᑎᓂᕐᒥᒃ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᑐᓴᖅᑕᕗᑦ.

- **ᐊᓕᓴ:** ᖁᔭᓐᓇᒦᒃ ᑲᑎᒪᔩᑦ ᐅᐸᓗᒥ ᐅᕙᑎᖕᓂᑦ ᑲᑎᒪᖃᑎᖃᕆᐊᖅᑐᓚᐅᕋᐸᓯ. ᐊᔾᔨᖕᒋᓐᓂᒃ ᐊᐅᓪᓚᖅᑎᑦᑎᕝᕕᒋᓂᐊᖅᑕᕗᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᑦ ᑕᑯᔭᒃᓴᐅᑎᐅᔪᒥᒃ ᐊᒻᒪᓗ ᐊᐅᓚᔪᒥᒃ. ᐊᒻᒪᓗ ᐊᐱᖅᖁᑎᒃᓴᖃᖅᑐᒍᑦ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐱᔪᒪᐸᓗᑕ.
- **ᕈᐊᐸᕐ:** ᐱᑦᑕᐅᔫᖅᖃᐅᔪᖅ. ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᓈᓴᐅᑏᑦ ᐱᓯᒪᔭᕗᑦ. ᐱᑦᑕᐅᔫᔪᖅ ᑐᑭᓯᐅᒪᔾᔪᑏᑦ ᐅᕙᑎᖕᓄᑦ.
- **ᒪᓄ:** ᑐᑭᓯᐅᒪᔾᔪᑎᑦᑎᐊᕙᐅᖅᖃᐅᔪᖅᑕᐅᖅ ᐅᕙᑎᖕᓄᑦ. ᖁᔭᓐᓇᒦᒃ.

- ᐃᓚᖕᒋᑦ ᐅᖃᐅᓯᐅᔪᑦ ᐆᒪᔪᓕᕆᔨᐊᓛᒃᑯᓐᓂᑦ ᒪᖃᐃᓐᓇᐃᑦ ᐅᐸᓗᖕᒋᑦ.
- **ᐊᓕᓴ:** ᐊᐅᓪᓚᕈᔾᔨᖃᑦᑕᖅᐱ?
- **ᕈᐊᐸᕐ:** ᖃᑉᓯᐊᕐᔪᐃᑦ ᐅᑭᐅᑦ ᐊᓂᒍᖅᑐᓂᑦ ᓇᓐᓄᓚᒃᓴᐃᑦ ᓄᖑᑕᐅᑲᐅᑎᒋᖃᑦᑕᓚᐅᖅᑐᑦ ᐅᑭᐅᒃᓵᒃᑯᑦ. ᓯᑯᓇᓵᖃᑦᑕᖅᑎᓪᓗᒍ, ᓇᓐᓄᓚᒃᓴᐃᑦ ᓄᖑᑲᐅᑎᒋᖃᑦᑕᓚᐅᖅᑐᑦ. ᓯᑯᔭᕌᖓᑦ ᓯᑯᑲᐅᑎᒋᖃᑦᑕᓕᖅᑐᑦ ᐊᒻᒪᓗ ᓇᓄᐃᑦ ᐊᐅᓪᓚᖅᓱᑎᒃ. ᓲᕐᓗ ᕿᓇᓗᒐᖅᑎᑐᑦ, ᕿᓇᓗᒐᖃᑦᑕᓚᐅᖅᑐᑦ ᐱᖓᓱᖓᓂᑦ ᐱᓇᓱᐊᕈᓯᕐᒥᑦ ᐊᑯᓪᓕᕈᕐᕕᖕᒥᑦ ᐊᑉᖁᓵᖅᓯᒪᔭᕌᖓᑦᑕ. ᐊᒥᕋᐃᔭᕐᕕᐅᓕᖅᑐᖅ. ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓ ᐱᔾᔪᑕᐅᔪᒃᓴᐅᔪᖅ. ᑐᒃᑐᐃᑦ ᐊᑉᖁᓵᖅᓵᓚᓂᖅᓴᐅᖃᑦᑕᓕᖅᑐᑦ ᐊᖅᖄᓂᒥᐅᖓᓂᑦ.

11:45-ᒥᑦ ᑲᑎᒪᓂᖅ ᐃᓱᓕᑦᑐᖅ

2021 AERIAL SURVEY OF THE WESTERN HUDSON BAY POLAR BEAR SUBPOPULATION

FINAL REPORT

November 16, 2022

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and Amélie Roberto-Charron[6]

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Disclaimer

The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment.

We dedicate this report to Markus Dyck; a dear friend and colleague who will be missed but of whom we have many fond memories. Your dedication to the future of polar bears and their place in the world will forever be remembered. We will do our best to make your contributions known to others. This is for you Markus. See you later mate.

(Photo credit: J. Ware, 2020)

SUMMARY

Relative to most polar bear subpopulations, the Western Hudson Bay (WH) subpopulation has been the subject of significant, long-term research, monitoring and management efforts. Concerns about the observed effects of long-term changes in sea-ice habitat, harvest pressure and increasing levels of human-bear conflict have made this subpopulation the focus of attention in discussions about the status of polar bears globally. Recently, part of the monitoring regimen for WH has involved a plan to conduct aerial surveys at 5-year intervals in-order to closely track abundance and distribution. Here, we report the third in this sequence of surveys.

Using double observer (sight-resight) and distance sampling methods from helicopter and fixed wing platforms, an aerial survey of the WH polar bear subpopulation was conducted in late August and early September 2021. Survey design, including study area stratification and transect spacing, was based on previous surveys in 2011 and 2016. Transects were oriented perpendicular to the coastline to align with bear density gradients.

We recorded a total of 194 bears in 125 groups. Like previous surveys, bears were concentrated along the coast and offshore islands, although both lone individuals and family groups were also regularly sighted inland, particularly within the Wapusk National Park region. The estimated abundance of WH in 2021 was 618 bears (SE=119.3, CI=425-899, CV=0.19). Comparison to aerial surveys estimates from 2011 and 2016 suggests that WH may be decreasing in abundance. Post-stratifying the results by sex and age classes revealed significant declines in the abundance of adult female and subadult bears between 2011 and 2021.

We were unable to definitively conclude whether the finding of declining abundance in WH over the last decade, specifically that of adult females and subadults, was the result of reduced survival and recruitment, movement of bears into neighbouring subpopulations (emigration), or harvest pressure. Based upon the multiple lines of evidence reviewed in

this report, it is plausible that all these factors have contributed to some degree. Of particular concern, however, is our finding that the observed declines are consistent with long-standing predictions regarding the demographic effects of climate change on polar bears. If these apparent trends continue, the progression of a reduced subadult cohort into the adult age class, combined with an already reduced adult female class, reproductive senescence, and mortality amongst older bears, may result in cascading effects on WH abundance and reproductive performance over the next decade. We therefore provide several recommendations for timely follow-up to these findings.

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1. INTRODUCTION

1.1. Status of Polar Bears

Across the Arctic, warming temperatures and changes in circulation patterns have led to profound changes in sea-ice, including declines in its extent and thickness, earlier spring melt (‘break-up’), later fall advance (‘freeze-up’) and longer seasonal ice-free periods (Stroeve et al. 2012; Overland and Wang 2013; Stern and Laidre 2016; Stroeve and Notz 2018). These climate-induced changes have been identified as the ultimate threat to the status of polar bears (*Ursus maritimus*); a species whose life history is reliant upon sea-ice as a primary habitat for foraging, movement, and reproduction (Wiig et al. 2015; Regehr et al. 2016). The sensitivity of polar bears to changes in sea-ice habitat quality is evident from the numerous field studies that have demonstrated associations between sea-ice habitat metrics and polar bear survival, body condition, growth, energy expenditure, movement patterns, distribution, reproductive performance and abundance (e.g. Stirling et al. 1999; Regehr et al. 2007; Rode et al. 2010; Cherry et al. 2013, 2016; Lunn et al. 2016; Obbard et al. 2016; Johnson et al. 2020; Laidre et al. 2020a and b; Pagano et al. 2021). Although most of these studies have documented negative consequences for polar bears from long-term changes in Arctic sea-ice, some have documented positive responses, in terms of body condition and abundance (Rode et al. 2014; 2018; Laidre et al. 2020b; Dyck et al. 2020a and b; Regehr et al. 2018; SWG 2016).

Observed variation in the responses of polar bears to a warming Arctic has largely been consistent with predictions (Stirling and Derocher 1993; Derocher et al. 2004; Stirling and Parkinson 2006; Stirling and Derocher 2012; Regehr et al. 2016). Subpopulations in southern portions of the species’ range, where the annual sea-ice melts completely during summer and autumn (e.g. polar bears in the seasonal sea-ice ecoregion *sensu* Amstrup et al. 2008) have been the first to experience negative effects (e.g. Striling et al. 1999; Rode et al. 2010; Obbard et al 2016; Lunn et al. 2016; Johnson et al. 2020). In contrast, some subpopulations in northern portions of the range are currently showing positive effects as these regions shift from multi-year sea-ice to thinner, annual sea-ice and a longer period of open water (Rode et al. 2014, 2018, 2021; Laidre et al. 2020b; Dyck et al. 2020a and b; Regehr et al. 2018). Thinner, annual ice provides better habitat for both polar bears and their ice-breeding seal prey. Combined with rising ocean temperatures, longer periods of open water, and greater penetration of sunlight into the water column, marine ecosystem productivity in these northerly areas appears to have increased thereby increasing the carrying capacity to support higher densities of bears (Stirling and Derocher 2012; Rode et al. 2014, 2018, 2021; Laidre et al. 2020; Dyck et al. 2020a ; Dyck et al. 2020b ; Häder et al. 2014, Frey et al. 2018). However, these benefits are predicted to be temporary and ultimately replaced by negative effects on polar bears if sea-ice conditions continue to deteriorate in the future. Overall, therefore, the predicted long-term consequence of continued sea-ice loss is that global polar bear abundance is likely to decline by greater than 30% over the next 4 decades (Regehr et al. 2016).

The polar bear is listed as a “Vulnerable” species by the International Union for the Conservation of Nature (IUCN) (Wiig et al. 2015) and a “Species of Special Concern” within Canada (ECCC 2020). Across its circumpolar range, the species is comprised of 19 subpopulations. Current assessments regarding the state of knowledge and demographic status of these subpopulations presents a varied picture reflecting the expected variation in subpopulation responses to climate change, as well as the immense financial and logistical challenges of monitoring this species in its remote, inhospitable range. As of 2021, ten subpopulations were assessed as ‘Data Deficient’, two as ‘Likely Increased’, four as ‘Likely Stable’ and three as ‘Likely Decreased’ by the IUCN’s Polar Bear Specialist Group (PBSG 2021).

1.2 Western Hudson Bay Subpopulation

The Western Hudson Bay (WH) subpopulation is one of three currently assessed as ‘Likely Decreased’ by the PBSG. This subpopulation ranks as one of the most intensively studied large mammal populations worldwide, with a history of scientific research and monitoring dating back five decades (e.g. Jonkel et al., 1972; Stirling et al., 1977;

Derocher and Stirling, 1995a; Regehr et al., 2007; Stapleton et al. 2014; Lunn et al. 2016; Dyck et al. 2016). Implemented predominately through mark-recapture sampling, findings from these studies suggest that WH abundance increased during the 1970s, remained somewhat stable, and then declined by an estimated 22% between 1987 and 2004 (Derocher and Stirling 1995; Lunn et al. 1997; Regehr et al. 2007). A more recent analysis suggests the population remained stable between 2001 and 2011 concurrent with a period of stability in sea-ice conditions (Lunn et al. 2016). In addition to these trends in abundance, linkages have been established between sea-ice conditions in WH and polar bear body condition, reproduction, movement patterns, distribution and survival (e.g. Stirling et al. 1999; Regehr et al. 2007; Cherry et al. 2013, 2016; Lunn et al. 2016; Johnson et al. 2020).

Long-term concern for WH polar bears, centers on the impact of a progressively earlier spring sea-ice break-up and delayed fall freeze-up that has, and will, result in bears spending longer periods on land where they have limited access to food (Stirling et al. 1999; Regehr et al. 2016; Stern and Laidre et a l. 2016; Johnson et al. 2020). Satellite-derived sea-ice data for the period 1979 to 2014, indicate that the length of the annual ice-free period in WH is increasing by approximately 9 days per decade (Stern and Laidre 2016). If this trend continues, further reductions in body condition, reproduction, survival and ultimately the abundance of polar bears are expected to occur (Molnar et al. 2010; Lunn et al. 2016). Harvest of WH bears is an additional concern that has the potential to exacerbate subpopulation decline if not managed appropriately.

Within Canada, the assessment of WH status based on scientific knowledge contrasts with that based on Indigenous Knowledge (IK); as summarized by several authors (McDonald et al. 1997; Dowsley and Taylor 2006; Tyrrell 2006; Nunavut Tunngavik Incorporated 2007; Committee on the Status of Endangered Wildlife in Canada 2008; Dowsley and Wenzel 2008; Canadian Wildlife Service 2009; York et al 2016; PBTC 2021). Canada's Federal-Provincial-Territorial Polar Bear Technical Committee (PBTC) assesses WH as "likely declined" based on scientific information and as "increased" based on IK (PBTC 2021). There is consensus between science and IK that polar bear abundance in WH has increased since the 1970's (Tyrell 2006). There is also agreement that polar bear distribution has changed, that more bears are being sighted around communities, that sea-ice breakup is occurring earlier, and that climate change is negatively influencing seal populations (NWMB 2007). However, in contrast to scientific evidence, Inuit perceptions of WH do not support the notion that abundance has declined since the mid-1980's (Tyrell 2006). Reports of more bears summering on land in the Nunavut portion of Hudson Bay (i.e., the Kivalliq region) and increased incidences of problem bears around camps and communities have been attributed to factors such as higher abundance, habituation of bears to human activities such as ecotourism, changes in behavior due to capture and handling for scientific research, and increasing use of unmanaged garbage dumps in communities along the Hudson Bay coastline (Stirling and Parkinson 2006; Dyck et al. 2007; NWMB 2007; Stirling et al. 2008a; Smith et al. 2022).

With the large body of scientific knowledge about observed and predicted declines in status and the disparities between science- and IK- based assessments, monitoring of WH has become a priority amongst governments, co-management agencies and stakeholders. Accurate and timely information is essential for detecting sudden changes in the subpopulation's status, if and when they occur (Von Graven et al. 2012; Derocher 2013); assessing population viability (e.g. Lunn et al. 2016); supporting adaptive or 'state-dependent' management measures, such as adjustments in harvest levels (Regehr et al. 2017a, b); and resolving apparently diverging views.

In 2011, the GN adopted the use of aerial surveys for monitoring WH (Stapleton et al 2014). Aerial surveys have been used for decades as a tool in monitoring wildlife species worldwide. Although their use for studying polar bears is a recent development, they have proven to be an effective means of monitoring subpopulation abundance, distribution and reproductive performance (Aars et al. 2009, 2017; Stapleton et al. 2014, 2016; Obbard et al. 2015, 2018; Dyck et al. 2017; Conn et al. 2021; Wiig et al. 2021). Aerial surveys are relatively inexpensive and quick compared to the mark and recapture methods that have typically been used to study polar bears. As such, they are an ideal tool for providing the frequent information updates needed to support near-real time adjustments in the management and conservation of a subpopulation undergoing unprecedented change. For communities in Nunavut that have expressed concern about the handling of bears or potential sampling biases in mark-recapture studies, aerial surveys have also offered an acceptable alternative. Unlike mark-recapture studies, however, aerial surveys do not provide estimates of demographic parameters such as survival rates, needed for population modelling and trend projection, nor do they allow sampling of individual bears to support studies in other areas of priority such as body condition, growth, diet, disease status, movements, habitat use and contaminant burdens (Von Graven et al. 2012). In the context of WH, therefore, aerial surveys and mark-recapture are both considered key components of the monitoring scheme.

A subpopulation wide aerial survey of WH was designed and implemented in 2011 (Stapleton et al. 2014). Based on the results and subsequent simulations, it was determined this survey should be repeated at 5-year intervals in-order to provide sufficient power to detect changes in subpopulation abundance (GN unpublished data). The survey was repeated in 2016 using similar methods (Dyck et al. 2017). Results from both surveys were used to facilitate status assessment by the IUCN and PBTC, as well as inform local harvest management decisions (PBSG 2021; PBTC 2021). Here we report the results of the third scheduled aerial survey of the WH subpopulation, conducted in 2021.

2. OBJECTIVES

The study’s objectives were to:

a) Generate an accurate and precise estimate of polar bear abundance in WH via aerial survey.

b) Evaluate the distribution of polar bears in WH during the 2021 ice-free season.

c) Compare results from the 2021 survey with those of surveys conducted in 2011 and 2016 to examine trends in subpopulation abundance, composition and distribution.

3. METHODS

3.1. Study Area

The annual range of the Western Hudson Bay (WH) subpopulation, located at the southern extent of the global polar bear distribution, stretches across roughly 435,000 km^2 of Hudson Bay and the adjacent coastline including portions of Nunavut, Manitoba, and Ontario (Figure 1). WH is part of the Hudson Bay complex that includes the neighboring Foxe Basin (FB) and southern Hudson Bay (SH) subpopulations (Obbard et al. 2010; Thiemann et al. 2008, Peacock et al. 2010). Although there is marked spatial overlap of polar bear movements from these three subpopulations while on the sea-ice (e.g., Stirling et al. 1999; Obbard and Middel 2012; Sahanatien et al. 2015), past capture-mark-recapture studies (Stirling et al. 1977; Derocher and Stirling 1990; Ramsay and Stirling 1990; Kolenosky et al. 1992; Taylor and Lee 1995; Derocher et al. 1997; Lunn et al. 1997, 2016), genetic studies (Paetkau et al. 1995, 1999; Crompton et al. 2008; Malenfant et al. 2016; Viengkone et al. 2020), and analyses of satellite telemetry data (Stirling et al. 1999; Obbard and Middell 2012; Sahanatien et al. 2015; Viengkone et al. 2020) generally support the currently accepted WH subpopulation boundary (Obbard et al. 2010).

Our study area has been well-described by Brook (2001), Dredge and Nixon (1992), Ritchie (1962), Clark and Stirling (1998), Peacock et al. (2010) and Richardson et al. (2005) and includes the areas described by Stapleton et al. (2014) and Lunn et al. (2016). The terrestrial portion of the study area stretches for approximately 1,500 km from about 35 km southeast of the Manitoba-Ontario border and northwards into Nunavut (approximately 20 km south of Chesterfield) (Figure 1). In addition to Rankin Inlet, the communities of Whale Cove and Arviat (Nunavut) and Churchill (Manitoba) are located within the boundaries of WH. In general, the southern portion of the study area displays the characteristics of the Hudson Plains ecozone and the Coastal Hudson Bay and Hudson Bay Lowlands. As described by Dyck et al. (2017), the northern portion exhibits Taiga and the Southern Arctic ecozone (Ecological Framework of Canada 2016). Where trees (black spruce [*Picea mariana*], white spruce [*P. glauca*], and tamarack [*Larix laricina*]) are quite common in the southern extents, dwarf birch (Betula nana), willows (*Salix spp*.), and ericaceous shrubs (*Ericaceae spp*.) are the norm to the north. The near-coastal southern areas exhibit elevated beach ridges, marshes and extensive tidal flats. There is very little relief (<200 m) with underlying continuous and semi-continuous permafrost. Sea-ice is absent in this region generally from July to November (Stirling et al. 1999; Scott and Marshall 2010; Stern and Laidre, 2016), and biting insects are plentiful during the summer (Twinn 1950).

Spatial separation of WH polar bears from individuals in neighboring subpopulations of the Hudson Bay complex is most complete during the late summer and early fall ice-free season when bears are on land (Peacock et al. 2010). Polar bears of WH come ashore when sea-ice levels diminish to ≤ 50% (Stirling et al. 1999; Cherry et al. 2013, 2016), which generally occurs during July (Stern and Laidre, 2016). Once on land, the bears segregate by sex, age class, and reproductive status within the study area where they exhibit fidelity to their terrestrial summer retreat areas (Stirling et al. 1977; Derocher and Stirling 1990). In general, adult males are found along the coastline, pregnant females and females accompanied by offspring are found in the interior denning area, which is mostly included within Wapusk National Park, and subadults are distributed throughout the study area (Stirling et al. 1977; Derocher and Stirling, 1990; Ramsay and Stirling 1990; Clark and Stirling 1998; Clark et al. 1997; Richardson et al. 2005; Stapleton et al. 2014). When sea-ice reforms during November, all bears except pregnant females return to the ice. Pregnant females give birth in terrestrial dens during December and early January, and these family groups generally depart their dens in March and April to return to the sea-ice (Jonkel et al. 1972; Stirling et al. 1977; Ramsay and Stirling 1988).

3.2. Survey Design and Field Methods

3.2.1. Survey Timing

The polar bear aerial survey was conducted in late August to early September 2021. This survey window during the ice-free period was selected because (a) all polar bears of the WH population are forced to be on land during this time; (b) range overlap between subpopulations within the Hudson Bay complex reaches a minimum, since polar bears exhibit a high degree of fidelity to terrestrial habitats during this period (Derocher and Stirling 1990; Lunn et al. 1997; Stirling et al. 2004; Parks et al. 2006); (c) the absence of ice and snow means that polar bears are readily observable against the landscape; (d) pregnant females are less likely to have begun denning yet and can be detected while moving towards their inland denning area (Stapleton et al. 2014); (e) non-denning bears have not yet begun to make directional northerly movements as they are known to do in the late fall, prior to freeze-up (e.g., Stirling et al. 1977; Derocher & Stirling 1990; Stirling et al. 2004); and (f) the two previous surveys in 2011 and 2016 occurred during a similar window.

3.2.2. Stratification

Like the 2011 and 2016 surveys (Stapleton et al. 2014; Dyck et al. 2017), we implemented a systematic, stratified study design to allocate sampling effort and improve estimate precision. For consistency, we used the same strata and sampling transects as the 2016 survey; themselves a modification of those used in the 2011 survey (Figure 2). The survey strata included the following derived polar bear density distributions: 1) very low, 2) low, 3) moderate, and 4) high. Descriptions of these strata, as provided by Dyck et al. (2017) are presented in Table 1. Polar bears tend to congregate along or near the shoreline during the ice-free season (Derocher and Stirling 1990; Towns et al. 2010), so overland transects were oriented roughly perpendicular to the coast (i.e., against the coastal density gradient; hereafter denoted as perpendicular transects) to improve precision and minimize potential biases (Figure 2; Buckland et al. 2001).

3.2.3. Aircraft

One de Haviland Twin Otter fixed-wing aircraft with radar altimeter and an Airbus AS 350B2 twin engine rotary-wing aircraft with radar altimeter were used to complete the survey. All aircraft throughout the survey maintained, as close as possible, an altitude of 400 feet above ground level (AGL) and an air speed of between 70 and 90 knots for the fixed-wing, and 70 to 80 knots for the rotary-wing aircraft while flying on transect. The Twin Otter fixed-wing aircraft was used to complete the low-density stratum within Nunavut and the very low and moderate density strata west and north of the high-density stratum bounded by the Churchill River, Manitoba, in the south. The twin engine fixed-wing and its ability to fly on one engine was chosen to increase safety while flying over extensive open water transects characteristic of the northern half of the survey study area within Nunavut.

3.2.4. Double Observers

The double observer pair (sight/resight) method is a variation of physical mark-recapture (Pollok and Kendall 1987). The aircraft's front and rear observers comprise two independent survey teams, visually 'marking' (i.e., front observers' sighting) and 'recapturing' (i.e., rear observers' resighting) polar bears. Observer teams must be independent to estimate detection probabilities. This resultant information provides an independent estimate of the number of bears present in the survey strip that were not observed by either team (Laake et al. 2008; Buckland et al. 2010).

The double observer pair method requires two pairs of observers on each of the left- and right-hand sides of the aircraft (Figure 3) (Buckland et al. 2001; Pollock and Kendall 1987). One "primary" observer sits in the front seat of the aircraft and a "secondary observer" is located behind the primary observer on the same side of the aircraft. To ensure visual isolation, a barrier was installed between same side observers to remove any visual cues that could modify an observer's ability to sight the animal. Observers waited until bear groups passed before calling out the observation to ensure independence of observations. The data recorder/recorders categorized and recorded counts of each bear (group) into “primary only”, “secondary only”, and “both”.

3.2.5. Fixed-Wing

Within the fixed-wing aircraft, we utilized an 8-person platform; 4 dedicated observers, 2 data recorders (for each of the left and right primary and secondary observer pairs) and a pilot and co-pilot. Observers within the fixed-wing survey crew included members of Hunters and Trappers Organizations in the Kivalliq region of Nunavut. The observers were further divided into primary and secondary teams, each isolated from the other using visual barriers between the seats as well audio barriers using two independent intercom systems monitored by each of a primary data recorder/navigator and a secondary data recorder/navigator. The pilot’s responsibilities were to monitor air speed and altitude while following transects pre-programmed on a Garmin 650T Geographic positioning system (GPS). The data recorder/navigators were responsible for monitoring a second and third identically programmed GPS unit for the purposes of double-checking the position as well as to record the geographic position, body condition, composition and numbers of observed polar bear groups on data sheets.

The positioning of the four observers within the aircraft differed during and between survey days and was adjusted to account for any possible variability in sightability or the detection of polar bears associated with seat position. The front and rear observer exchanged spots mid-day, and the left and right sets of observers changed periodically. The data recorders and pilots did not vary their position within the aircraft and remained consistent during the survey.

3.2.6. Rotary-Wing

The AS 350B2 only allowed for a four-person configuration due to weight and balance issues while carrying full fuel as well as seating configuration. Using this configuration only the secondary observers were dedicated observers while the left primary observer seat was occupied by a data recorder/observer and the right primary position by a pilot/observer. Additionally, observers could not exchange primary and secondary positions using this configuration to determine sightability differences between seating positions. While the methods used during this study generally followed those used by Stapleton et al. (2014), it is important to note that no pooling of front and rear observers was made. All observations made during this study were independent.

3.2.7. Distance Sampling

In addition to the deployment of the double observer pair method within all aircraft, we also collected observations using distance sampling. The distance sampling method followed Buckland et al. (1993, 2004, 2010) and used the *mrds* (Laake et al. 2012) R package (R Development Core Team 2009) to model stratified line transect observation data and estimate density and abundance for polar bears. Using the conventional distance sampling approach (CDS), we modeled the probability of detecting a group of polar bears and their densities within five delineated strata as a function of distance where the detection function represents the probability of detecting a group of polar bears, given a known distance from the transect (Buckland et al. 2001). Recognizing that other variables may affect the detection probability, density estimates were also derived using multiple covariate distance sampling (MCDS), which allowed us to model probability of detection as a function of both distance and one or more additional covariates (Buckland et al. 2004). This approach was explored in-order to increase the reliability of density estimates made on subsets of the data based on terrain, vegetation, and environmental conditions, and to increase precision of the density estimates within each unique density-derived stratum (Marques et al. 2007).

For the fixed-wing portion of the survey only, and in addition to flying to the observed bears for position and data collection, we also used distance bins marked out with streamers and tape on the wing struts after Norton-Griffiths (1978) (Figure 4). In total, 6 distance bins were used including the following: 0-200 meters, 200-400 meters, 400-600 meters, 600-1,000 meters, 1,000-1,500 meters, and 1,500-2,000 meters. Though binned observations were not used during analysis, they did inform on the precision of binning for distance sampling platforms when compared to the actual observation waypoint recorded.

3.2.8. Observations

Polar bears observed while flying along a transect line were considered on-transect while those observed while ferrying to, from, or between transects, or to bear and/or wildlife sightings, were considered off-transect. Because polar bears are often found in groups, each observation (whether an individual or group) represented a group of polar bears. In this work a group of polar bears was defined as one or more individuals within a visually estimated 100-meter radius of one another. All observations were investigated by moving off the transect line to the center of the group as they were initially observed, to record the location, group size, sex/age classes, body condition.

We determined sex and body condition, to the extent possible, from approximately 30 meters altitude. Sex of bears was determined based on body size, the presence of morphometric characteristics (e.g., such as scars, large head, thick neck, long fur on front legs, vulva patch and urine stains) and behavior when encountered (SWG 2016). Age-class assessment from the air can be accomplished reliably for adult males, pregnant females, and members of family groups (Government of Nunavut, unpublished data; SWG 2016). Based on these methods, polar bears were classified as male or female, and as adult males (6+ years), adult females (5+ years), subadult males (2 to 5 years), subadult females (2 to 4 years), yearlings (>1 and < 2 years), and cubs-of-the-year (<1 year).

A general, relatively robust though subjective fat index has been successfully used in past studies to assess body condition of polar bears (Stirling et al. 2008; SWG 2016; Laidre et al. 2020a, b; Dyck et al. 2020a, b; Dyck et al. 2022). Standardized body condition indices [i.e., poor (1), fair (2), good (3), excellent (4) and obese (5)] were scored for each individual bear (Stirling et al. 2008). Each aircraft had at least one experienced biologist on board who could identify age classes and body conditions of observed bears with confidence.

Additional covariates that could affect detection probabilities were recorded for each group including activity when first sighted, height and density of surrounding vegetation, habitat class, visibility, cloud cover, glare and general habitat description (Table 2). Observation times were kept to a minimum to reduce disturbance and stress. All distances to the observations were measured perpendicularly from the transect line to the center of the observation and recorded along with the observation's date and time of day. It was assumed that the bear location at initial observation was determined accurately with no effect of movement after detection on estimated distance from the transect line. The distance from line was then estimated using shapefiles of transect lines with GIS methods. All aircraft deployed the distance sampling methods and collection of covariate data consistently across the study.

3.2.9. Coastal Contour Transects

Like the 2011 WH survey (Stapleton et al. 2014), we additionally surveyed along comprehensive coastal contour transects covering the entire coastline of WH, independent of the transect flown perpendicular to the coast. Contour transects were flown at or slightly below the high-water line (HWL) with one side of the aircraft dedicated to monitoring tidal flats and near-shore waters (i.e. swimming bears). We surveyed along coastal contours as close to high tide as possible to minimize tidal flat exposure and reduce the need to double-back to ensure that the coastal zone was comprehensively covered. Observers looked for bears as far as they could reasonably see, not within a pre-defined strip width. Because perpendicular transects were extended to the shoreline and over tidal flats (where applicable), some bears along the shoreline could be sighted from both perpendicular and coastal transects. Bears sighted on tidal flats or in nearshore waters were considered within the coastal zone (i.e., on land, where area could be estimated with GIS) in order to calculate density. Although collected, coastal contour data are not analyzed and present in this report since such data were not collected during the 2016 survey thus preventing comparison of the three surveys.

3.3. Data Analyses

3.3.1. Data Screening and Truncation

Data was initially screened for outlier observations that occurred at far distances therefore creating a tail on the detection function that can be difficult to fit. A right truncation distance that eliminated the upper 5% of observations was considered to minimize the influence of these observations (Buckland et al. 1993, Stapleton et al. 2014). The blind spot under each aircraft was estimated using geometric formulas. From this, left truncation distances were estimated for the twin otter as 98.9 m and approximately 70 m for the AS350B helicopter. Adjusted distance from the transect line was then estimated as the distance from the transect line minus the left truncation distance for each aircraft.

Like the survey in 2016 (Dyck et al. 2017), but unlike that in 2011 (Stapleton et al. 2014), we left truncated both the front (pilot and data recorder) observations from the helicopter rather than only left truncating the rear observations. The rationale for this was that we wanted to keep the data sets as similar as possible for the double observer analysis. There were 3 observations of 7 bears that were only observed in the rear observer blind spot by the front observers in the helicopter. Therefore, the degree of reduction due to left truncation of the helicopter data was not large.

3.3.2. Distance Sampling Double Observer Analysis

Analysis methods

Mark-recapture distance sampling methods were applied to the survey data (Buckland et al. 2004, Laake et al. 2008a, Laake et al. 2008b, Buckland et al. 2010, Laake et al. 2012). A mark-recapture / distance sampling model assuming point independence was used which allows estimation of the detection probabilities at the transect line (or left truncation distance) using independent double observer methods with distance sampling methods used to model the decline in sighting probabilities as a function of distance from the survey line.

Covariates that affected bear sightability were considered that included environmental, observer and survey factors (Table 3). These covariates included group size, aircraft type, observer, and visibility. Like the 2016 survey, a remote sensing-based covariate (*RSveg*) based on LANDSAT 8 vegetation classification was also considered (Figure 5). The rationale behind this covariate was that it would systematically index dominant vegetation types in the proximity of observations therefore providing the best comparison of habitat and potential obstruction of observations across all observations. The main categories in Figure 5 that were present in the study area were gravel, shrub, trees, low vegetation, and water.

The twin otter fixed-wing aircraft had 2 dedicated observers per side of the aircraft. The A-star helicopter had 2 dedicated surveyors in the back seat of the helicopter and the pilot and data recorder/navigator as observers in the front. The pilot and data/recorder did not have the same view as the observers and were distracted by piloting the helicopter and navigating/data recording. Therefore, special covariates were formulated for the pilot and data recorder/observers in this aircraft.

Distance model fitting

A sequential process was used for model building. First, parsimonious distance sampling models were formulated using a mark recapture model with constant detection probabilities. Once the most supported distance model was determined, parsimonious mark-recapture models were formulated using the most supported distance model as a base model in the mark-recapture model analysis. As a final step, optimal distance and mark-recapture models were combined and assessed for goodness of fit and overall parsimony. Information theoretic methods (Burnham and Anderson 1992) methods were used to assess relative model fit. More exactly Akaike Information Criterion (AIC) were used as an index of model parsimony with lower scores indicating a model that explained the most variation in the data set with the least number of parameters. The difference between the most supported model and given model was evaluated (ΔAIC) to indicate relative support with models with ΔAIC values of less than 2 being of interest. Akaike weights were used to estimate proportional support of models. Models were averaged based on AICc weights using the *AICcmodavg* (Mazerolle 2016) package in program R (R Development Core Team 2009). The AIC score indexes relative fit but does not provide a test of overall goodness of fit. Goodness of fit tests incorporated the *mrds* package as well as graphical methods were used to further evaluate fit of the most supported models.

Abundance estimates were derived for the most supported models with variances being estimated assuming sequential systematic sampling (the S2 estimator in *mrds*). This estimator accounted for sequential lines sampling in the survey (Innes et al. 2002, Fewster 2011).

3.3.3. Trend Analyses

Given the previously observed declines in WH abundance and predictions regarding future decline, we examined estimates from the current series of 3 aerial surveys that have been conducted at 5-year intervals for evidence of decline. Data from the previous 2 projects were re-analyzed using the same methods (Dyck et al 2017). Trends in polar bear abundance estimates from the 2011, 2016 and 2021 WH distance sampling surveys were initially compared graphically. Estimates of trend were then derived using ratios of estimates. A simulation approach that assumed log-normal distributions of estimates was used to test for significance between successive estimates as well as confidence limits on overall (gross) change and yearly change in estimates. Log-normal distributions were assumed since they best describe the distribution of estimates from distance sampling and are also assumed when estimating confidence limits of estimates (Buckland et al. 2004). One thousand simulations of estimates were generated from a log-normal distribution for each year. The proportion of simulations where gross change (the ratio of successive estimates i.e. $N_1:N_2$) was greater than 1 was tallied. If this proportion was less than 0.05 then a significant decline was suggested. We note this test is equivalent to a one tailed test between two abundance estimates testing for evidence of decline (null hypothesis H_o: $N2 \geq N_1$ and alternative hypothesis H_a: $N_2 < N_1$). Confidence limits were then derived based on the 2.5th and 97.5th percentile of the resulting distributions of gross (GC) and annual change (with $\lambda = GC^{(1/\text{survey interval})}$). The 1-tailed hypothesis provides a more powerful test for decline than a 2-tailed test (which tests if the 2 estimates are equal). A 1-tailed test was justified given previously observed and predicted declines in the subpopulation; a question of key management interest.

To explore whether change was occurring within specific segments of the subpopulation, abundance estimates from 2011, 2016, and 2021 were post-stratified to derive estimates for adult males, adult females, and subadults. Trends within these segments were examined. Finally, there was interest in assessing whether change was occurring evenly across the study area or whether there were geographic patterns in change. To explore this, the 2011, 2016, and 2021 study areas were divided into Nunavut (Area 3), the area from Nunavut to the Nelson River (Area 2), and from the Nelson River to eastern boundary of WH in Ontario (Area 1). To do this required dividing the moderate strata used in 2011 and 2016 into a moderate north and south strata as was used in 2021. Estimates were derived based on strata in these areas for overall estimates as well as estimates of age/sex groups. The 2011 strata were redrawn and the areas of the 2 new strata double checked to make sure they were similar to the original single stratum.

Distance sampling analyses were conducted using the *mrds* (Laake et al. 2012) R package (R Development Core Team 2009). Data was explored graphically using the *ggplot* (Wickham 2009) R package with GIS analyses conducted using the simple features (sf) (Pebesma 2018) R package and QGIS program (QGIS Foundation 2020).

4. RESULTS

4.1. Survey Timing and Effort

Extensive forest fires across Canada in the summer of 2021 limited the availability of suitable aircraft and delayed the planned start of 2021 WH polar bear survey by approximately 1 week. The survey was flown between August 21st and September 6th. This compares to survey windows of August 13th to 29th and August 12th to 21st in 2011 and 2016, respectively. Using 50% sea-ice cover in WH as an index of ice break-up (e.g. Laidre and Stern 2016), the 2011, 2016 and 2021 surveys were conducted at 67 to 83, 55 to 64 and 65 to 81 days post-break-up, respectively.

Including weather-related delays, 2021 survey strata between Chesterfield Inlet and Churchill, flown using the Twin Otter, took 6 days to complete (August 21-27). The remainder of the study area, flown by helicopter, took 13 days (August 24-September 6). Approximately, 41 and 72 hours were flown with the Twin Otter and helicopter, respectively, for an estimated total distance of approximately 17,000 km, including ferry time. Weather during the survey was good and allowed for completion of all transects and coastal contour surveys.

4.2. Summary of Observations

Overall, 194 bears in 125 groups were observed during distance sampling with 176 observed on transect and 18 observed off transect (Table 4). Group sizes ranged from single individuals up to groups of 6. Eighteen cubs-of-the-year (COY) and 17 yearlings were observed. Mean litter sizes were 1.46 (SD: 0.50; n = 13) and 1.39 (SD: 0.63; n = 13) for COYs and yearlings, respectively[1].

Polar bear sightings were not uniformly distributed across WH (Figure 6; Table 4). Similar to surveys in 2011 and 2016, the highest concentrations of bears were documented in the high-density stratum, encompassing the historical Environment Canada study area, including Wapusk National Park and along the coast of southeastern WH. Eight percent

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[1] Includes off transect observations

of observations were in Nunavut. In general, observations were concentrated along or near the coast throughout the subpopulation (Figure 6). However, inland bears >10 km from the coastline were often recorded in the high-density stratum and less frequently observed in the southeastern portion of WH (the southern moderate density strata). Adult males were concentrated along the coast. In contrast, lone adult females or adult females with offspring, either COY or yearlings, were most often observed inland.

4.3. Abundance Estimation

4.3.1. Truncation of Observations

Observation data were left and right truncated (based on 97th percentile of distance from line after adjustment for left truncation) (Table 5 and Figure 7) yielding a dataset of 154 bears for analyses. Left truncation was based on measured blind spots for each aircraft and eliminated data not available to both observers. Using the right truncation distance (2100 m) eliminated excess observations at the tail end of the detection histogram that would exert influence on fitting of detection functions. A sensitivity analysis was conducted to validate the right truncation distance (Appendix 1).

4.3.2. Covariates

Habitat classes (as classified by observers) did not have sufficient sample sizes to allow modelling of detection functions for each class. These were pooled into similar categories (Figure 8). Overall, the detection histograms for each category were relatively similar (when considering differences in sample sizes). The remote sensing vegetation classification (RSveg) was also pooled into 3 categories (Figure 9). The detection histograms for each category were also relatively similar (when considering differences in sample sizes). When plotted on a map, both of these pooled vegetation classifications corresponded to shoreline and inland habitats (Figure 10). Other descriptors of vegetation, recorded for each bear observation, were vegetation height and density. Low density vegetation generally corresponded with vegetation of low height (Figure 11).

Vegetation density also corresponded to general habitat class with the shore/tidal areas mainly having low vegetation class ratings (Figure 12).

Group size can influence detectability. However, the relative range of group sizes was small with most observations being of single bears and few large groups over 3 bears (Figure 13). A bear's activity when first observed was also considered. Observations were pooled into 4 main categories of activity (Figure 14). Visibility was recorded, with the majority of observations 101 of 102 being in clear to broken conditions and only one observation in fog. Glare, based on sun angle, was also recorded with 91 of 102 observations having no glare. Sun angle was overhead for most observations.

4.3.3. Summary of Double Observer Data

Sample sizes for observers were much higher for the helicopter that flew the higher density strata. Detection probabilities ranged from 0.6 to 1 (Table 6). Detection histograms amongst individuals were relatively similar for the helicopter-based observers (Figure 15). Low sample sizes precluded assessment of individual histograms for the Twin Otter-based observers.

A related issue occurred with the helicopter where the pilot and a potentially weaker observer were on the right side of the aircraft for the majority of the survey (Figure 16). This potentially resulted in a reduced number of observations on this side of the aircraft. To assess this issue, models were fit to exactly test the difference in detection function and double observer probabilities of sighting on the line for the right side of the helicopter. A HeliSide covariate was used which modelled detection functions (distance sampling) or detection probabilities at the line (MR analysis) for the right and left side of the helicopter and fixed-wing (sides pooled given limited sample sizes). Results of this analysis, presented in appendix 1, found little evidence of differences in detection function between the left and right side of the helicopter.

4.3.4. Model Selection

The first phase of model selection was assessment of covariates (Table 3) that describe the shape of the distance sampling detection function (Table 7). None of the covariates had substantial support with a constant hazard rate model being most supported. A variety of models had some support as indicated by ∆AICc scores of less than 2. The next step was assessment of variation in sightability near the aircraft from double observer models. The most supported distance model (hazard rate constant) was used for this analysis. Of covariates considered, vegetation density (VegDensity) had the highest support (Table 8).

The most supported distance and double observer models were combined (Table 9). Minimal variation was detected in the detection function based on covariates. A constant distance sampling model showed the highest support with double observer detection varying with vegetation density. Other models with helicopter navigator/data recorder and habitat class as covariates were supported, however, the sensitivity of abundance estimates (N) to model variation was low.

Goodness of fit of model 1 (Table 9) to the distance sampling (χ^2=4.7, d.f.=4, p=0.31), mark-recapture (χ^2=15.3, d.f.=12, p=0.23) components was adequate with adequate overall fit of (χ^2=20.1, df=12, p=0.23). A Von-Mises test also suggested adequate fit (Test statistic=0.028, p=0.98). Plots of fit to the model 1 suggest reasonable fit of predictions to front (observer=1) and rear (observer 2) observations as well as duplicate observations (seen by both observer) (Figure 17). Also suggested is minimal dependence between observer detections (as suggested by no directional trends in histograms by distance which indicate conditional probabilities). The mean single observer probability at the line for model 1 was 0.69 (SE=0.04, CV=0.07) with a combined (both observers) probability of 0.89 (SE=0.04, CV=0.46). Figure 18 shows predictions from model (DS (HR(.), MR(VegDensity) with predictions further delineated by VegDensity category and also group size observed, as represented by data point size

Estimates of abundance from the most supported model (Table 9, model 1) by strata are given in Table 10. A model averaged estimate of abundance using the models in Table 9 is 618 bears (SE=119.3, CI=425-899, CV=0.19). Of additional interest was a breakdown of estimates by adult males, adult females and subadults. Post-stratified estimates for these groups are presented in Table 11.

4.4. Abundance Trend Analyses

Data sets from WH aerial surveys conducted in 2011 and 2016 were analyzed and the results compared with those of the 2021 survey. A visual comparison of distance sampling total abundance estimates suggests a decrease in abundance from 2011 to 2021 (Figure 19). Additionally, post-stratified estimates suggest a decrease in the adult female and subadult classes with no apparent trend in adult males (Figure 19).

Gross and annual changes were estimated using the ratio of survey estimates with confidence limits calculated assuming a log-normal distribution of estimates for adult males, adult females, subadults, and all bears (Table 12). The annual rate of growth changed from 0.90 for subadults to 1.00 for adult males for the period 2011-21. The estimate of gross change was significantly lower than 1 for the 2011-21 interval for females, subadults, and all bears (at $\alpha=0.1$) suggesting declining abundance. This estimate was based on the proportion of log normal simulations where the estimate from 2021 was greater than the estimate of 2011. Similar results were derived from standard t-test comparisons of estimates (Appendix 2).

Of further interest was geographic trends in each sex and age class. Downward trends in adult females and subadults were suggested in Area 2 with less distinctive trends in other areas (Figure 20). Trends were significant for adult females and subadults between 2011 and 2021 for Area 2 but were not significant in other areas (Table 13).

4.5. Polar Bear Mortalities

Three polar bear carcasses were found while conducting the 2021 WH survey (Figure 21). Based on body size and dentition, one appeared to be an adult female and another a subadult. The third was identified as a 21-year-old adult male of known age, based on previous capture and tagging records. All three carcasses were in advanced stages of necrosis making cause of death difficult to determine. However, the adult male was noted to be in poor body condition making starvation a potential contributing factor in this case. A fourth polar bear carcass, that of an 18-year-old adult female, was found in WH in August 2021 during polar bear research unrelated to the aerial survey (S. Atkinson pers. obs). In 2 of the 4 cases, polar bears were observed feeding on the carcasses of their conspecifics.

5. DISCUSSION

5.1. Distribution

With some exceptions, discussed later in this report, the summertime distribution of polar bears within WH in 2021 (Figure 6) was broadly similar to previous surveys in 2011 and 2016(Stapleton et al. 2014; Dyck et al. 2017). The northern part of WH extending north from the Manitoba-Nunavut boundary, referred to as the low-density strata or Area 3 in this study, contained relatively few bears. Eight percent of bear observations recorded during the survey were found in this region, along the coast and offshore islands, compared to 6% and 5% in 2011 and 2016, respectively (Stapleton et al. 2014; Dyck et al. 2017). In the area extending Northwards from the Nelson River up to the Nunavut border, referred to as Area 2 in this study, the highest densities of bears occurred along the coastline. However, we also encountered a significant number of individuals far inland (>10 km), mostly within the bounds of Wapusk National Park. In contrast, virtually all polar bears in the region from the Nelson River eastward into Ontario (denoted as Area 1) were highly concentrated in a relatively narrow strip along the coast. In both areas, adult males were most often found on the coast while adult females tended to occupy areas further inland. These patterns of distribution and segregation have been well-documented in previous studies and attributed to several factors, including variation in the availability of suitable inland habitats for denning, the avoidance of conspecifics, thermoregulation, and insect avoidance (Stirling et al. 1977; Derocher and Stirling 1990; Lunn et al. 1997; Clark and Stirling 1998; Richardson et al. 2005).

Looking at estimates of abundance for the 3 areas of WH (Figure 20), we also found little evidence that the distribution of bears within WH has changed significantly over the last 10 years. Although, the estimated abundance of adult female and subadult bears in Area 2 decreased significantly between the 2011 and 2021 surveys, concurrent increases of these types of bears in Areas 1 (Cape Tatnum) or 3 (Nunavut) of WH were not found (Table 13). This suggests that the observed declines are not the result of more bears occupying regions to the north or southeast of what has historically been considered the core summer range of the WH subpopulation. Emigration, reduced survival and/or reduced reproductive performance could account for the observed declines. However, the finding that bear numbers (in absolute terms) were unchanged in areas bordering the northern (Area 3) and southeastern (Area 1) boundaries of the subpopulation between 2011 and 2021, makes emigration a somewhat less plausible explanation for the observed declines. Increased emigration from WH, if driven by changes in habitat availability, quality or phenology, such as sea-ice break-up patterns, would likely be concurrent with, or preceded by, apparent reductions in the abundance of bears in these ‘boundary’ areas. This is especially the case in Area 3, bordering the Southern Hudson Bay subpopulation, where there are high densities of bears and the distribution of remnant summertime sea-ice is known to affect the location bears come ashore (Stirling et al. 2004; Cherry et al. 2013).

Given our findings regarding distribution and the low number of bears observed and estimated in the Nunavut (Area 3) portion of the subpopulation, on-going mark-recapture studies in WH that focus sampling effort in a core study area centered around Churchill and Wapusk National Park (Area 2) are unlikely to contain significant bias in estimates of abundance or vital rates due to unsampled bears in Nunavut. However, relative to historical mark-recapture sampling (e.g. Regehr et al. 2007; Lunn et al. 2016), increased sampling effort is recommended in the area east of the Nelson River (Area 1), along the coast towards Ontario. As documented by the 2011, 2016, 2021 aerial surveys, and coastal surveys conducted by the Government of Manitoba, this area is typically occupied by several hundred bears during the on-land period. In generating estimates of WH abundance for conservation and management planning, it has been assumed by this and past aerial survey studies (Stapleton et al. 2014; Dyck et al. 2017) that these bears are part of the WH demographic unit. Studies are currently in progress to test this assumption (D. McGeachy per comm.).

5.2. Abundance

The estimated total abundance of WH polar bears in 2021 was 618 bears (SE=119.3, CI=385-852) based on model averaging. Overall, there was minimal change in estimates with different models suggesting that this estimate is robust to model selection uncertainty. The relative simplicity of the most supported distance sampling (constant) and double observer (vegetation density) models was surprising, given the range of covariates included in the analyses. However, inspection of histograms from the observed data does not indicate a large degree of variation in detection functions beyond differences in sample sizes. The covariate explaining variation in detectability in the vicinity of the aircraft was vegetation density. Vegetation density was also associated with vegetation height whereby low-density vegetation tended to be of low height (Figure 8). Support for this covariate suggests differences in detection between the inland versus coastal areas of WH. Bears occupying inland shrub or treed habitats tended to be harder to detect than those in open coastal areas.

Similar to the WH 2021 survey, previous aerial survey-based estimates of abundance in both WH and the neighbouring Southern Hudson Bay (SH) subpopulation (in 2011 and 2016) have all relied on relatively simple models with vegetation density, or a similar covariate reflecting vegetation density and height, included in top models (Stapleton et al. 2014; Dyck et al. 2017; Obbard et al. 2015, 2018). These earlier surveys have also included a covariate describing visibility, based on weather conditions, in some top models. This covariate was not among top models in the 2021 analysis potentially due to good survey conditions (i.e. clear, sunny skies) as well as the reduced sample sizes relative to 2011 and 2016 surveys. Overall, despite some differences in aircraft types and observers, the similarity of models amongst the 2011, 2016 and 2021 surveys suggests the sampling methods, maintained across these surveys, are robust and yield comparable datasets suitable for long-term trend monitoring.

The precision of the 2021 estimate was comparable to that of previous WH aerial surveys with a coefficient of variation (CV) of 19% compared to 17% for both the 2011 and 2016 surveys. Aerial surveys of polar bears conducted during the on-land or minimal sea-ice seasons have proven to be a cost-effective monitoring tool in subpopulations where flat terrain and high densities of bears that show interannual fidelity to on-shore regions facilitate detections (Stapleton et al. 2014, 2016; Dyck et al. 2017; Obbard et al. 2015, 2018). These types of surveys have yielded abundance estimates with CVs ranging from 11-19%. In contrast, surveys over sea-ice during the spring tend to be more expensive and have resulted in CVs ranging from 13-39% (Macdonald et al. 1999; Wiig and Derocher 1999; Evans et al. 2003; Aars et al. 2009, 2017; Conn et al. 2016; Wiig et al. 2021). The relatively poor precision of some 'on-ice' surveys is due to low bear densities and reduced detection probabilities on ice. On-ice surveying of WH is not recommended for several reasons including potential cost, expected poor precision and the extensive range overlap amongst individuals of the WH, Foxe Basin and Southern Hudson Bay subpopulations that is known to occur on the sea-ice (Peacock et al. 2010; Viengkone et al. 2018). Continued monitoring of the WH subpopulation via summertime aerial surveys is recommended.

5.3. Assumptions and Potential Biases

Generating unbiased (accurate) abundance estimates via the distance sampling method used in the survey is dependent on several assumptions (Buckland et al. 2001). To satisfy the assumption that bears were randomly distributed with respect to distance from the transect line, we surveyed with systematically spaced transects oriented perpendicular to the coastal density gradient; the same or similar transects flown in previous surveys (Stapleton et al. 2014; Dyck et al. 2017). A second assumption is that all bears present on the transect line or at the point of left truncation (i.e. distance zero) when the aircraft flies over are detected (Buckland et al. 2001). This assumption was evaluated in our analysis by estimating the combined double observer detection probability. Combined detection at the point of left truncation was 0.89 in 2021 compared to 0.97 and 0.90 for the 2011 and 2016 surveys respectively. This suggests that most, but not all, bears 'on the line' were detected. However, this bias was corrected by estimated detection probabilities on the line using the mark-recapture distance sampling approach.

A third assumption is that bears are not disturbed/displaced from their initial location, by the approaching survey aircraft, before being observed. This potential source of bias was minimized by maintaining rapid flight speeds of up to 150 km/hr thus reducing opportunities for bears to move great distances before detection. Ninety-three percent of bears were either sitting, laying, standing, walking or swimming and 7% were running when first observed during the survey. This suggests that most bears did not move or did not move significantly in response to aircraft. Accurate measurement of the distance between a bear sighting and the transect line is also essential (Buckland et al., 2001). We used GPS and GIS to measure distance from the line in accordance with accepted methods (Marques et al. 2006) that have been used extensively for polar bear aerial surveys (e.g. Aars et al. 2009, 2017; Obbard et al. 2016, 2018; Stapleton et al., 2014, 2016; Wiig et al. 2021). Assuming bears were recorded at their initial location, as discussed above, this method should have provided accurate distance data.

Another potential source of bias was the difference in number bear observations between the left and right sides of the helicopter either due to chance or weaker observers on the right side that were not accounted for by the mark-recapture distance sampling models. Although relatively small samples may have limited power, additional analyses to explore this issue did not provide evidence of bias in abundance estimate (Appendix 1). Additionally, any potential bias, while affecting the overall abundance estimate would not explain differences in abundance trends amongst adult males, adult females and subadults that we observed between 2011 and 2021.

Polar bears that were entirely hidden from both front and back observers during the survey would not have been incorporated into the abundance estimate. Two sources of this ‘availability bias’ were possible in our survey. Some WH polar bears, typically pregnant females, use earthen dens during the ice-free season, entering them as early as August (Jonkel et al. 1972; Stirling et al. 1977; Clark et al. 1997; Clark and Stirling 1998; Lunn et al. 2004; Richardson et al. 2005). Although we cannot correct for bears that were underground and entirely unavailable for observation during the survey, the number of such individuals was likely low. During the survey we observed numerous dens, including some that were freshly excavated. When observed, these dens were inspected from the air but none were found to be occupied. Bear occupying dense vegetation may also be harder to observe from the air than those in more open habitats. This reduced probability of observation was accounted for in the abundance estimate by incorporating a vegetation density covariate in models. Nevertheless, some individuals sheltering under dense vegetation, such as willows or trees, may be completely unavailable for observation. In particular, inland areas parts of WH, are tree covered creating the potential for bears to be unobservable from the air. However, densities of bears in these areas tend to be low since these treed areas are not preferred habitat for most bears. Thus, while we are unable to quantify this potential source of availability bias, our impression was that although trees and brush impeded detection and reduced sighting probabilities, it is likely that very few bears on or near the transect line were completely concealed by vegetation.

Directional, or migratory, movements of bears in, out or within the study area during a survey could lead to under and over counting of bears or observations of the same individuals more than once. Polar bears in WH make several directional movements during the on-land period. The first of these involves their migration from sea-ice onto land in the summer, the timing of which is known to be determined by sea-ice concentration (Cherry et al. 2013; Cherry et al. 2016; Pilfold et al. 2017). To minimize this potential bias, the survey was timed to occur after bears had migrated to land. Using 50% sea-ice cover as an index of break-up (Laidre and Stern 2016), the 2021 survey occurred 65 to 81 days post-break-up. Although the survey start was delayed due to availability of aircraft, the survey window was almost identical to that of the 2011 survey (67 to 83 days post-break-up) but approximately 2 weeks later than the 2016 survey. All three surveys occurred well after the 50% sea-ice threshold and when Canadian Ice Service maps indicate that WH was essentially ice-free. Thus, all WH bears should have been on-land at the start of these surveys. WH bears are also known the make directional movements northwards later in the fall (Stirling et al. 1977, Derocher and Stirling 1990; Bohart et al. 2020). Again, the survey was timed to avoid this period thus eliminating potential for this source of bias. Finally, field work was completed within a narrow temporal window and the aircraft were able to cover large expanses of land within a single day. Therefore, distributional shifts within WH during the study period did not impact our results.

In summary, because this study met analytical assumptions and potential sources of bias were minimized, we believe the aerial survey-based estimate of 618 bears (SE=119.3, CI=385-852) accurately reflects the number of polar bears within the bounds of WH during August 2021. Any biases in the aerial survey would likely result in an underestimate of the true polar bear abundance in WH. However, we note that such bias, if present, would not affect the ability to detect trends in abundance since the same methods (hence the same biases) were utilized in the 2011, 2016 and 2021 surveys.

5.4. Trends in Abundance

Estimates derived for the WH subpopulation indicated a possible decline in total bear abundance between 2011 and 2021. The 2011 survey produced an estimate of 949 bears (95%CI: 618–1280), the 2016 survey an estimate of 842 bears (95% CI: 562–1121) and this survey derived an estimate of 618 (SE=119.3, CI=385-852) for 2021. Although differences amongst these estimates were not statistically significant, total abundance has declined consistently between successive surveys. Significant reductions in the number of subadult bears and adult females appear to account for this observed change. Interestingly, the abundance of adult males remained unchanged during this period. These changes may be the result of internal demographic processes within WH (i.e. rates of birth and death) and/or changes in distribution leading to increased emigration of bears out of WH and into neighbouring subpopulations on a temporary or permanent basis. As discussed below, a review of multiple lines of evidence provides support for both hypotheses.

Several lines of evidence suggest that internal demographics have played a role in the observed decline in WH subpopulation abundance. The finding that abundance of subadult bears and adult females has declined whilst that of adult males has remained stable is particularly striking given its consistency with both long-standing hypotheses and field studies. Reduced recruitment and survival of subadults, hence reduced abundance, are typically among the first demographics effects to occur within large mammal populations experiencing density-dependent regulation (Fowler 1987). For polar bears, it was first predicted almost 30 years ago that negative impacts from things such as climate change would first appear amongst subadult and adult female bears (e.g. Stirling and Derocher 1993; Stirling et al. 1999; Derocher et al. 2004; Robbins et al. 2012; Klappenstein et al. 2020). These two classes of bears have nutritional ecologies that are likely to make them more vulnerable to deteriorating environmental conditions than other polar bears. Subadult bears must sustain the energetic costs of growth whilst also gaining experience in hunting. Adult females have the added costs of repeatedly raising litters of offspring over periods of up to 2 years, which is predicted to reduce their tolerance of fasting relative to that of adult males (Robbins et al. 2012; Stirling and Derocher 2012).

A series of field studies have validated the hypothesis that subadult and adult female polar bears are more sensitive environmental conditions than other classes of polar bears such as adult males. In some subpopulations, experiencing long-term declines in sea-ice, reductions in body condition (itself a precursor to reduced survival) have been greater and/or more readily detected amongst the adult female and subadult classes (Obbard et al. 2006, Rode et al. 2010; Laidre et al. 2020). In WH, Johnson et al. (2020) found that body condition and energy metrics have declined over time in relation to earlier sea-ice breakup with the most significant effects seen amongst solitary adult females and subadult males. Studies, including several in WH, have also demonstrated that links between sea-ice conditions and survival are stronger amongst subadults and adult females relative to other age and sex classes (Regehr et al. 2007; Bromaghin et al. 2015; Lunn et al. 2016). For subadults in WH, declining abundance between 2011 and 2021 may also be the result of reduced recruitment. Aerial surveys in 2011 and 2016, found low numbers of the yearlings compared to other subpopulations suggesting that recruitment into the subadult age class was poor in these years (Stapleton et al. 2014; Dyck et al. 2017). Additionally, mark-recapture sampling during this period also suggests recruitment has been low in multiple cohorts with the number of yearlings, expressed as a proportion of total captures, being less than 0.06 in 6 of the 10 years (ECCC unpublished data). A series of years with poor recruitment from the yearling age class, combined with potentially lowered survival amongst subadult cohorts, may thus have contributed to the observed decline in subadult abundance.

Changes in distribution leading to increased emigration of bears from WH into neighbouring subpopulations, such as SH, could also account for some of the variation in abundance observed across the 3 aerial surveys. Prevett and Kolenosky (1982) found significant interannual variation in aerial counts of bears along the southern coast of Hudson Bay in Manitoba and Ontario, around the WH and SH boundary, during the ice-free period. They attributed this to ice-dependent variation in on-shore arrival locations suggesting that in years when bear counts on the Manitoba (or WH) side of the boundary were high counts on the Ontario (or SH) side tended to be low and vice-versa. A series of subsequent studies utilizing mark-recapture, telemetry and coastal survey data did not find evidence to support this hypothesis instead finding that WH bears exhibited a high degree of fidelity to on-shore areas during the ice-free period (Derocher and Stirling, 1990; Kolenosky et al.,1992; Lunn et al., 1997; Stirling et al., 1999; Stirling et al. 2004). More recently, however, analyses of telemetry data for adult females in WH have found that the timing of bears' movements to shore and the locations where they arrive on-shore are primarily influenced by environmental variables including wind direction, ice concentration and ice distribution during break-up (Cherry et al. 2013, 2016; Pilfold et al. 2017; Bohart et al. 2020). In particular, Cherry et al. (2013) found that on average approximately 12% of WH adult females came ashore outside the boundaries of WH, typically further south and east, and within the boundaries of the SH subpopulation. Additionally, WH bears were more likely to come ashore outside WH in years when there was more remnant summer-time ice in SH relative to WH (Cherry et al. 2013). Based on these findings, a degree of year-to-year variation in aerial survey-based estimates of WH should be expected as a result of ice-dependent variation in the locations bear come ashore. In years when ice remains longer in SH relative to WH, a higher proportion of WH bears may come ashore within the bounds of SH where they would be included in a SH rather than WH estimate.

Several lines of evidence support the hypothesis that the observed decline in WH abundance between 2011 and 2021 was to some degree the result of interannual variation in the distribution of bears between WH and SH. Aerial surveys of the SH subpopulation have been conducted concurrently with the surveys in WH in 2011, 2016 and 2021. Pooling estimates from the WH and SH surveys show a decline in total ‘Hudson Bay’ abundance from 2011 to 2016 but no change from 2016 to 2021 (Figure 22). Notably, between 2016 and 2021, the estimated abundance of SH increased by 223 bears while that of WH decreased by 224 (Northrup et al. 2022). Changes in both subpopulations, at least between 2016 and 2021, could therefore be accounted for by movement of WH bears into SH. Preliminary results from genetic mark-recapture work conducted along the coast of WH and SH provide additional evidence to support this hypothesis. In 2021, biopsy darting conducted as part of a genetic mark-recapture program found that 22% of bears sampled in SH had been previously sampled in WH only (McGeachy et al. 2022). In contrast, sampling within WH did not detect any recaptures of bears previously sampled in SH only.

Although they provide evidence of a potential distribution shift, comparison of abundance estimates for WH and SH, as well as preliminary findings of the genetic mark-recapture program should be interpreted with caution for several reasons. First, recent physical and genetic mark-recapture sampling effort has been markedly greater in WH relative to SH. For example, for genetic mark-recapture alone, more than 350 bears were marked within WH between 2017 and 2020, while none were marked in SH. This disparity in marking effort would have increased the likelihood of recapturing bears, originally marked in WH, within SH in 2021 and reduced the likelihood of SH marks being recaptured in WH. Second, the vast majority of bears marked in WH and recaptured in SH in 2021 were adult males yet the observed decline in WH appears to be due to declining numbers of subadults and females rather than adult males. Third, adult male bears are likely to exhibit greater flexibility in where they come ashore to spend the summer. Unlike adult females they do not require access to suitable denning habitat or in-land areas in which to avoid infanticide of dependent off-spring by conspecifics (Derocher and Striling 1990; Stirling et al. 2004). Consequently, trends in abundance resulting from sea-ice related shifts in summer range are more likely to be observed in WH adult males. Our findings that adult female and subadult abundance has declined while adult male abundance has remained unchanged are thus inconsistent with a range-shift hypothesis. Fourth, as noted above, WH bears exhibit a greater likelihood of coming ashore in SH in years when there is more remnant summer-time ice in SH relative to WH (Cherry et al. 2013). If an ice-dependent shift in summertime range were responsible for the observed decline in WH, sea-ice data for 2011, 2016 and 2021 should show that greater amounts of late break-up sea-ice were present in SH, relative to WH, in 2021 versus 2011 or 2016. However, the data suggest sea-ice that remnant sea-ice conditions in July, for example, were very similar in 2021 and 2011 (Figure 23). Finally, if trends in WH abundance were due to a distributional shift with bears moving out of WH, this shift would likely also be apparent within the boundaries of the subpopulation itself. We examined trends in abundance within 3 zones of WH and found declining abundance within the central or core zone of the WH summer range (zone 2). However, similar trends were not apparent in the areas of WH bordering the neighbouring SH and FB subpopulations. Again, these findings do not support a distribution shift hypothesis to explain the changes in WH abundance.

Harvest mortality has also likely contributed to the observed changes in WH abundance. Dependent on sea-ice conditions, the most recent demographic assessment of WH projected a long-term population growth rate ranging from 0.97 to 1.02 from 2011 onwards. Between 2011-2021, annual harvest in WH increased in accordance with the regulated Total Allowable Harvest (GN unpublished data; Figure 24) from a rate of approximately 2.5% of estimated abundance in 2011/12 up to 6.6% in 2021[2]. Since the rate of harvest has exceeded projected population growth, a decline in total abundance between 2011 and 2021 would be expected due to harvest pressure. However, the apparent decline in adult females and subadults but lack of trend in adult males is harder to rectify with a harvest-induced effect. Harvest in WH has been sex-selective with an annual average of 66% males between 2011 and 2021. Subadults have compromised approximately 26% of annual harvest [3], a level comparable with other subpopulations (GN unpublished data). Lunn et al. (2016) found that probability of harvest (*H*) was highest amongst young adult males (5-9 yrs) and lowest amongst adult females at 0.73 and 0.05, respectively. For subadults, H was 0.44 and 0.28 for males and females, respectively. Given the higher harvest pressure on adult males compared to other classes, a harvest-induced change in subpopulation composition would be expected to appear first amongst adult males rather than subadults and adult females, a pattern not consistent with our results.

5.5. Reproductive Performance

As discussed earlier in this report, aerial surveys in 2011 and 2016 suggested that reproductive performance in WH was poor. Relative to neighbouring subpopulations in the seasonal ice ecoregion, the number of yearlings, expressed as a proportion of all individuals observed, was low suggesting poor recruitment into the subadult age classes (Table 14). Similarly, mark-recapture sampling yielded relatively low numbers of yearlings in 6 of 10 years from 2011 to 2020 (ECCC unpublished data). These findings

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[2] Based on 2011 and 2021 aerial survey abundance estimates.

[3] Data available for the period 2010-2019 only.

suggest that over the previous decade WH has experienced a series of years with low recruitment into the subadult age class.

Results from the 2021 survey indicate that yearling numbers were higher and more comparable with levels seen in other subpopulations (Table 14). A similar improvement in yearling numbers was also seen in SH in 2021, suggesting conditions in Hudson Bay over the last few years have been generally favorable for raising offspring (Northrup et al. 2022). Whether this improvement in reproductive performance will continue is unknown. Regehr et al. (2015) estimated conditions necessary for polar bear population persistence which included recruitment levels of 0.1-0.3 yearling per adult female and adult female survival rates between 0.93-0.96. Raw aerial survey observation data indicate that recruitment was approximately 0.14, 0.15 and 0.31 in 2011, 2016 and 2021, respectively suggesting that reproductive performance although variable has likely been sufficient. Of concern with respect to future reproductive capacity in WH, is the apparent decline in subadult abundance. A reduced subadult cohort will eventually result in cascading negative effects on reproduction as these individuals enter the adult age classes and current adult bears begin to either enter reproductive senescence or die (Regehr et al. 2021).

5.6. Mortalities

What appears to be an unusually high number of polar bear carcasses were found in WH during 2021. Three were located during the aerial survey and 1 during other polar bear research activities. Based on flying times for both projects, this equates to approximately 3 carcasses per 100 hours of search effort. By comparison, while flying more than 1200 hours conducting aerial sampling of polar bears during the ice-free period in 4 other subpopulations in the seasonal ice ecoregion, over the last decade, 3 polar bear carcasses (0.25 per 100 hours) were found (S. Atkinson pers. obs.).

Cause and timing of mortality was unknown in all cases, although starvation was likely involved in one case. Interestingly, three of the 4 were adults which is somewhat unexpected since adult survival rates are higher than other age classes. The observation of other bears feeding on the carcasses is consistent with previous reports of cannibalism amongst polar bears (Lunn and Stenhouse 1985; Taylor et al. 1985; Amstrup et al. 2006; Dyck and Daley 2002; Derocher and Wiig 1999; Stone and Derocher 2007). Amongst bear species, polar bears display the highest reported levels of cannibalism (Allen et al. 2022). Amstrup et al. (2006) suggested incidences of predation and cannibalism amongst polar bears may reflect increased nutritional stress within a subpopulation. However, for the cannibalism observed in WH in 2021, it is unknown whether the mortalities were the result of conspecific predation or opportunistic scavenging on available carcasses.

Given the high number of carcasses found in 2021 and the fact that at least 3 were adults, careful documentation of future carcass observations is recommended to assess whether deeper investigation is warranted.

6. CONCLUSIONS

Ultimately, it is important to stress that we are unable to definitively conclude whether the finding of declining abundance in WH over the last decade, specifically that of adult females and subadults, is the result of reduced survival and recruitment, movement of bears into neighbouring subpopulations (emigration), or harvest pressure. Based upon the multiple lines of evidence reviewed in this report, it is plausible that all these factors have contributed to some degree. Of particular concern, however, is our finding that the observed declines in subadults and adult females are consistent with long-standing predictions regarding the order in which negative demographic effects will emerge amongst the different sex and age classes of polar bears due to climate related environmental change. If these apparent trends continue, the progression of a reduced subadult cohort into the adult age class, combined with an already reduced adult female class, reproductive senescence, and mortality amongst older bears, may result in cascading effects on WH abundance and reproductive performance over the next decade. We therefore recommend follow-up on these findings in several forms as follows:

1. Analyses of on-going physical and genetic mark-recapture programs are best suited to address the uncertainties arising from our aerial survey findings and are a strongly recommended next step in the assessment of WH. An analysis of physical and genetic mark-recapture data for 2011 to 2021 is needed to determine whether demographic effects such as reduced recruitment and survival of subadults have occurred in WH.

2. Management agencies should consider increasing monitoring efforts in WH by changing the frequency of future aerial surveys from every 5 years to every 3 years, in the near term. The purpose of this increased frequency would be to determine whether the subpopulation is entering a period rapid change in abundance requiring more frequent adjustments in harvest management strategy than at present.

3. Questions remain about the interannual movements of bears between WH and SH and the effect these movements have on aerial survey abundance estimates. It is also unclear whether the significant number of bears, mostly adult males, that occupy the southeastern coast of WH during the ice-free season are available for harvest by WH communities or whether these bears function as either an unharvested segment of the subpopulation or are in fact harvested in SH. Results from current genetic sampling and telemetry studies in this region are expected to address these questions and inform WH management. Priority should be placed on completing these studies.

4. A harvest risk assessment should be conducted to determine the impact of recent and future harvest under current conditions.

5. The high number of polar bear carcasses found during the WH survey is concerning. We recommend closer monitoring and reporting of polar bear carcasses found in WH during future aerial surveys and on-going mark-recapture studies.

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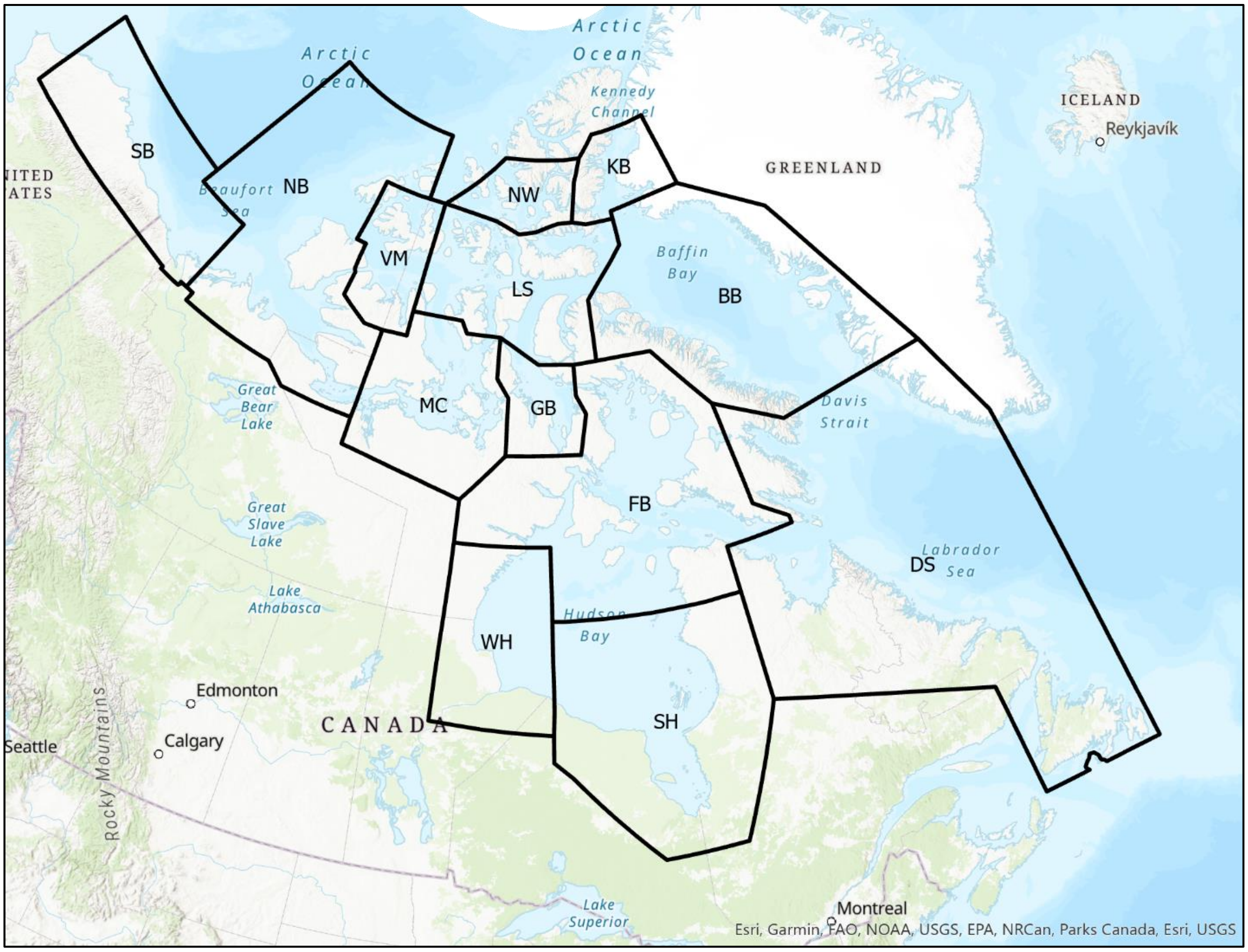

Figure 1. Boundaries of polar bear subpopulations that are partially or totally under management from Canadian jurisdictions. These include Southern Beaufort Sea (SB), Northern Beaufort Sea (NB), Viscount Melville Sound (VM), McClintock Channel (MC), Lancaster Sound (LS), Norwegian Bay (NW), Kane Basin (KB), Baffin Bay (BB), Gulf of Boothia (GB), Foxe Basin (FB), Davis Strait (DS), Western Hudson Bay (WH) and Southern Hudson Bay (SH).

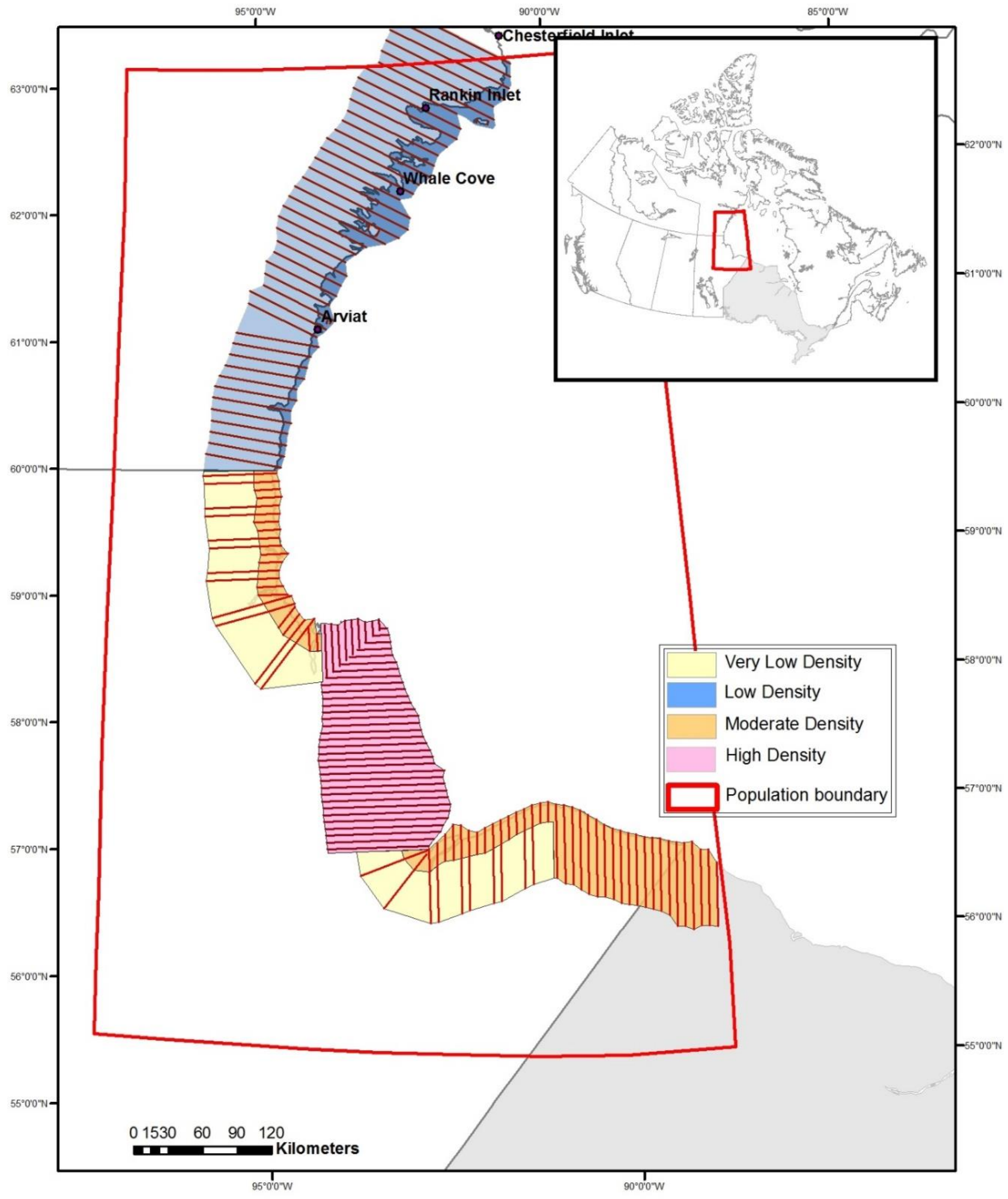

Figure 2. Survey strata and transects for the 2021 aerial survey of the Western Hudson Bay polar bear subpopulation.

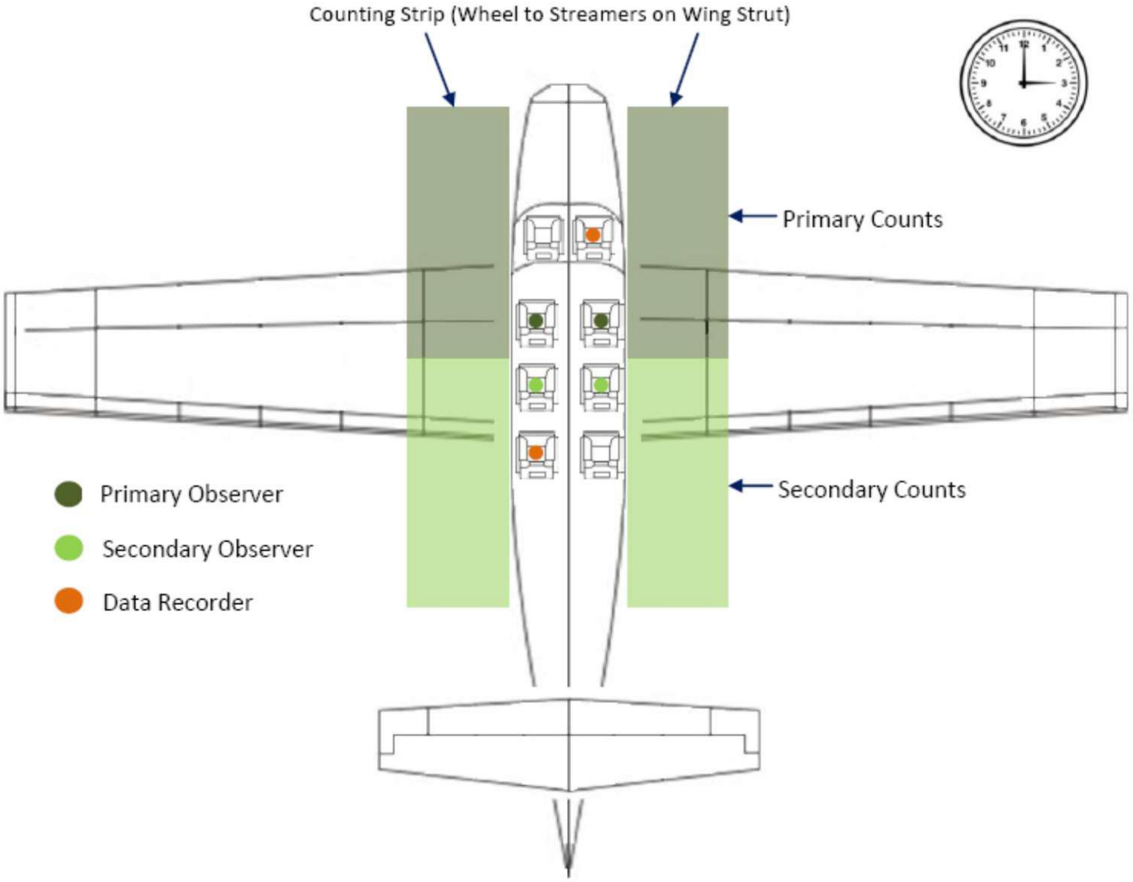

Figure 3. Observer position for the double observer method employed on this survey. The secondary observer calls polar bears not seen by the primary observer after the polar bear/bears have passed the main field of vision of the primary observer at a point halfway between same side primary and secondary observers. The small hand on a clock is used to reference relative locations of polar bear groups (e.g. “Polar bear group at 3 o’clock” would suggest a polar bear group 90o to the right of the aircrafts longitudinal axis).

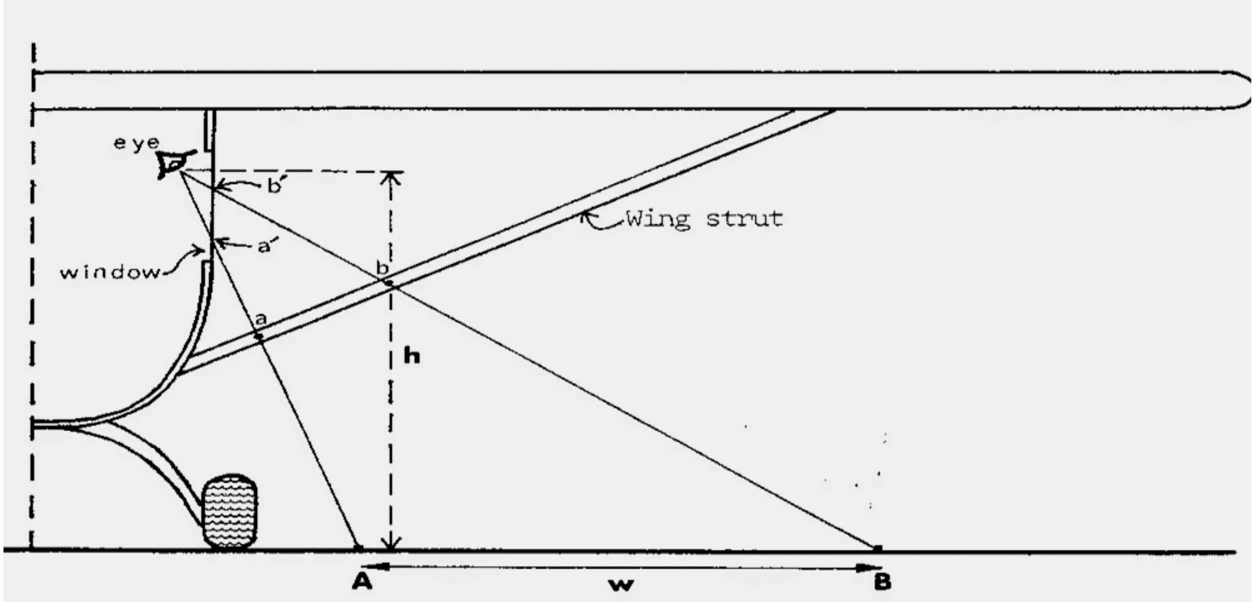

Figure 4. Schematic diagram of aircraft configuration for strip width sampling (Norton-Griffiths, 1978). W is marked out on the tarmac, and the two lines of sight a' – a – A and b' – b – B established. The streamers are attached to the struts at a and b, whereas a' and b' are the window marks. (After Jolly 1969)

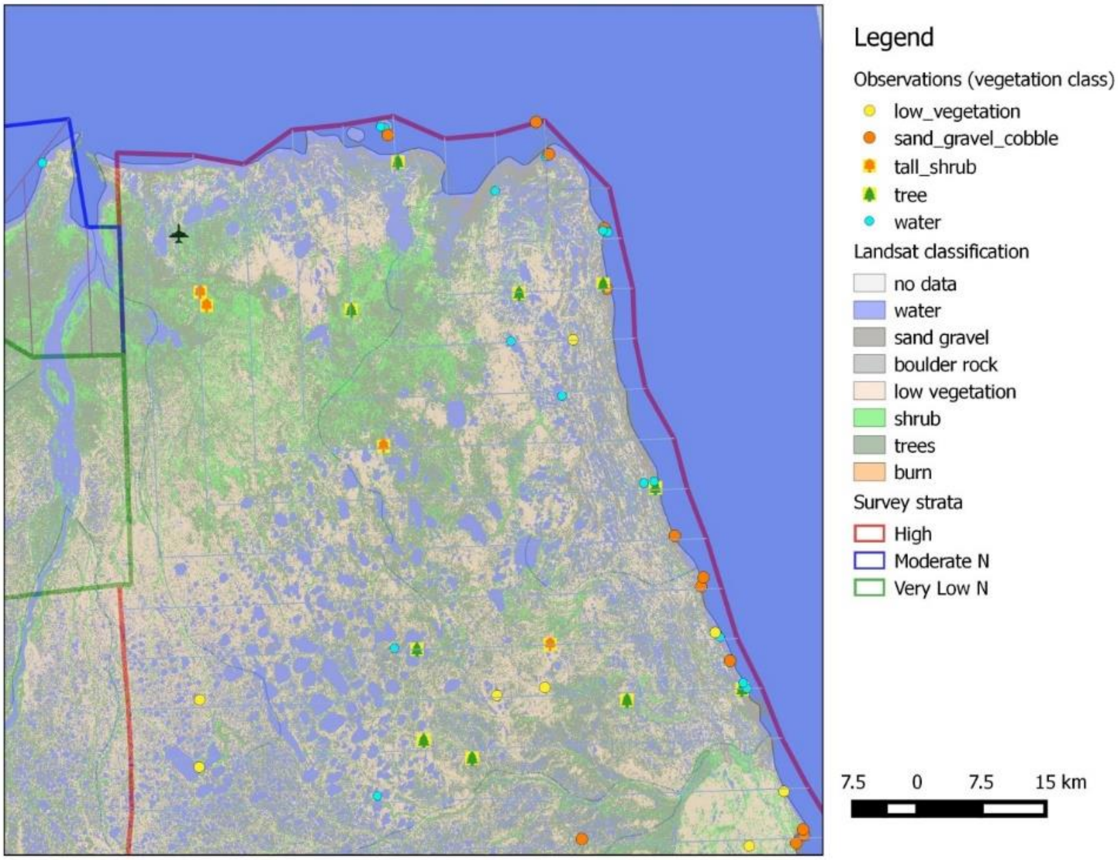

Figure 5. Landsat habitat classification and observations for a section of the 2016 high stratum (Dyck et al. 2017)

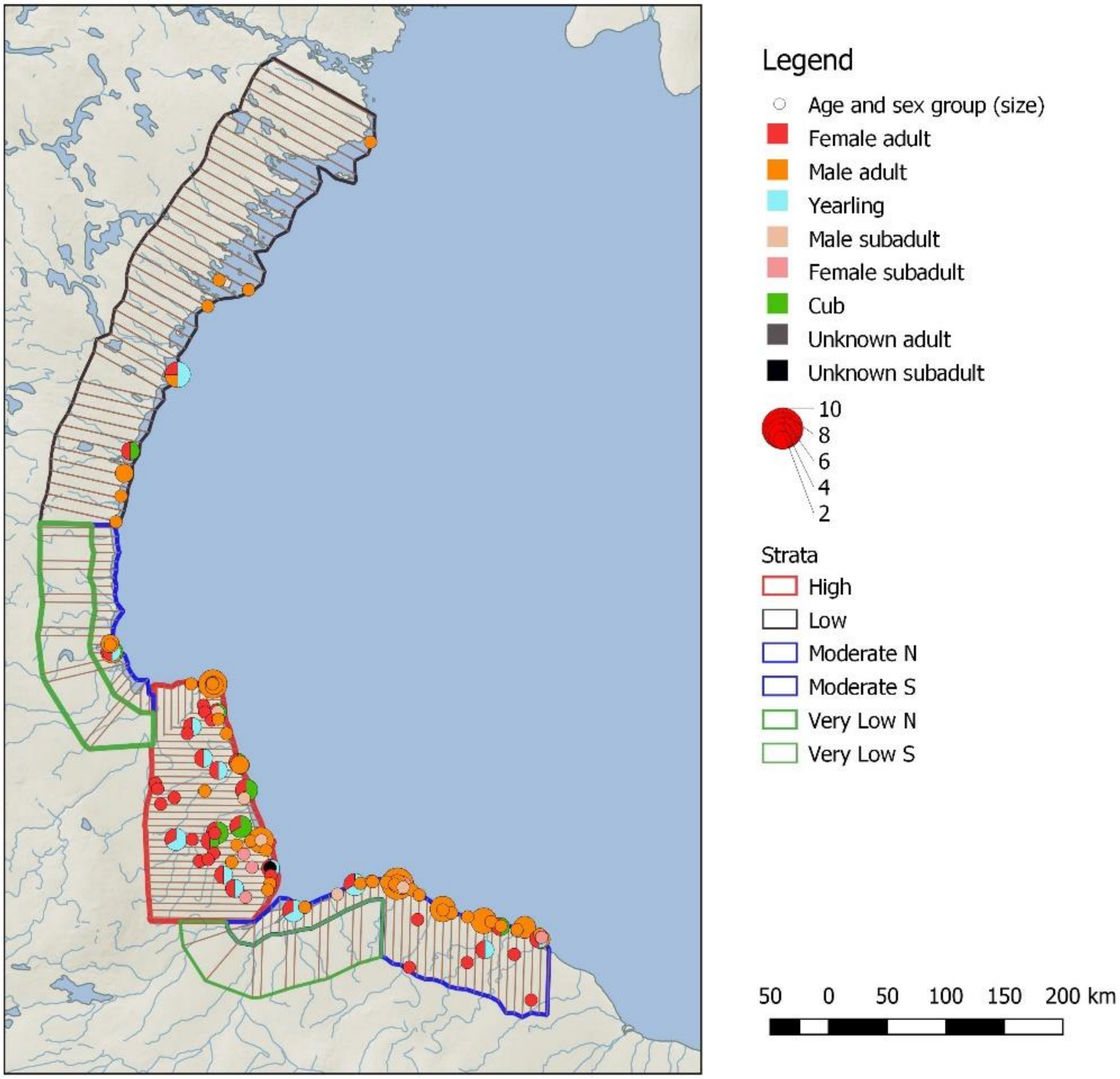

Figure 6: Distributions of bears observed on transect during the 2021 WH aerial survey; with group composition and size noted.

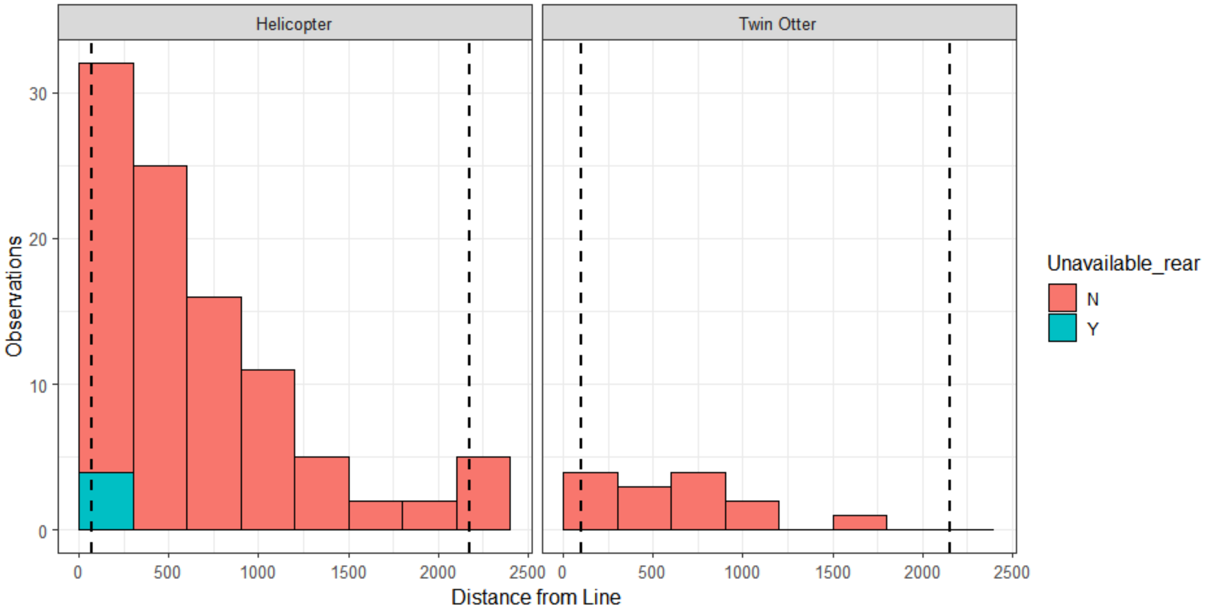

Figure 7: Detection histogram of full data set before right and left truncation by aircraft type. Left and right truncation distances are shown as hashed vertical line

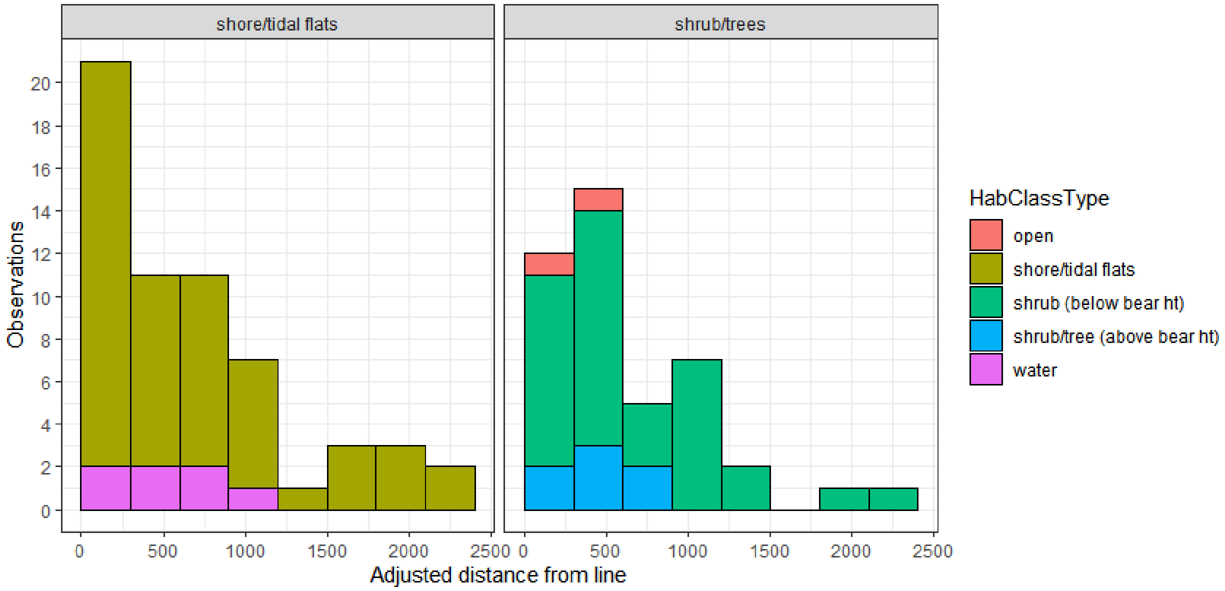

Figure 8: Detection histograms for pooled habitat class (HabClassP) categories with original categories shown as sub-bars. Adjusted distance from line (left truncation distance subtracted) is displayed on the x-axis.

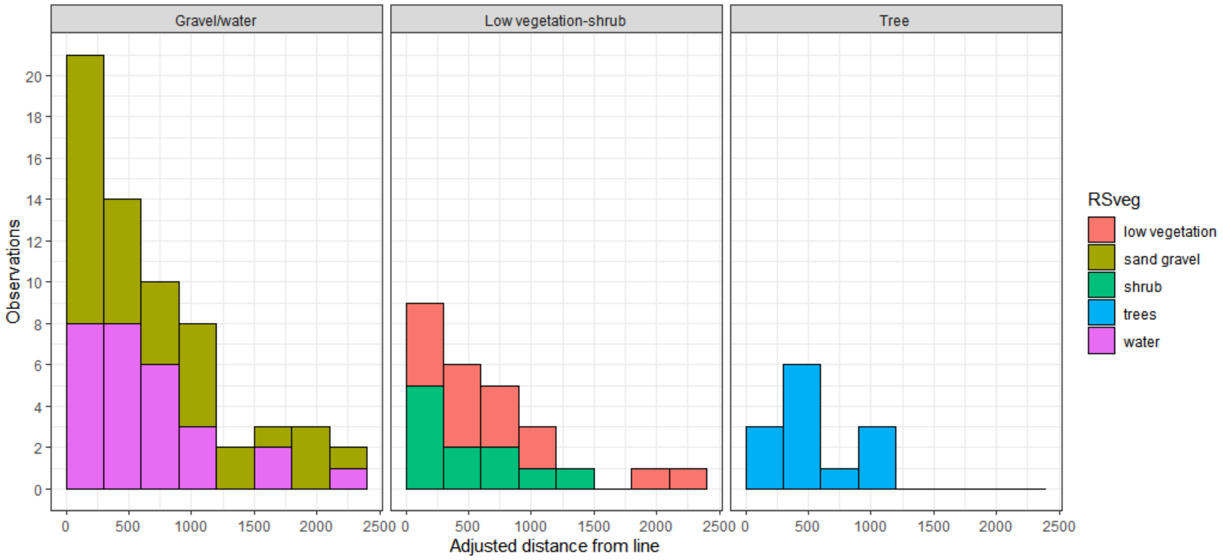

Figure 9: Detection histograms for pooled RSveg habitat categories with original categories shown as sub-bars. Adjusted distance from line (left truncation distance subtracted) is displayed on the x-axis.

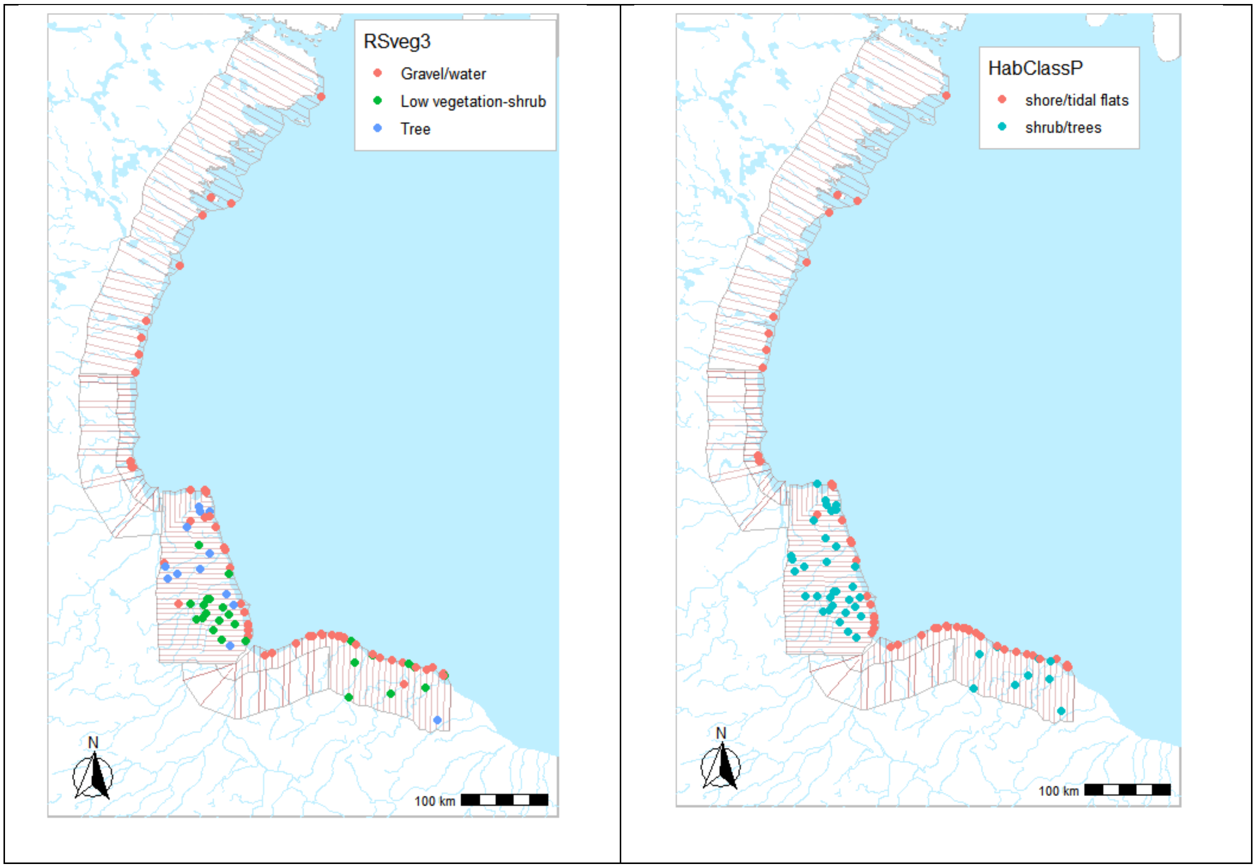

Figure 10: Distribution of observations by RSVeg and HabClass classifications.

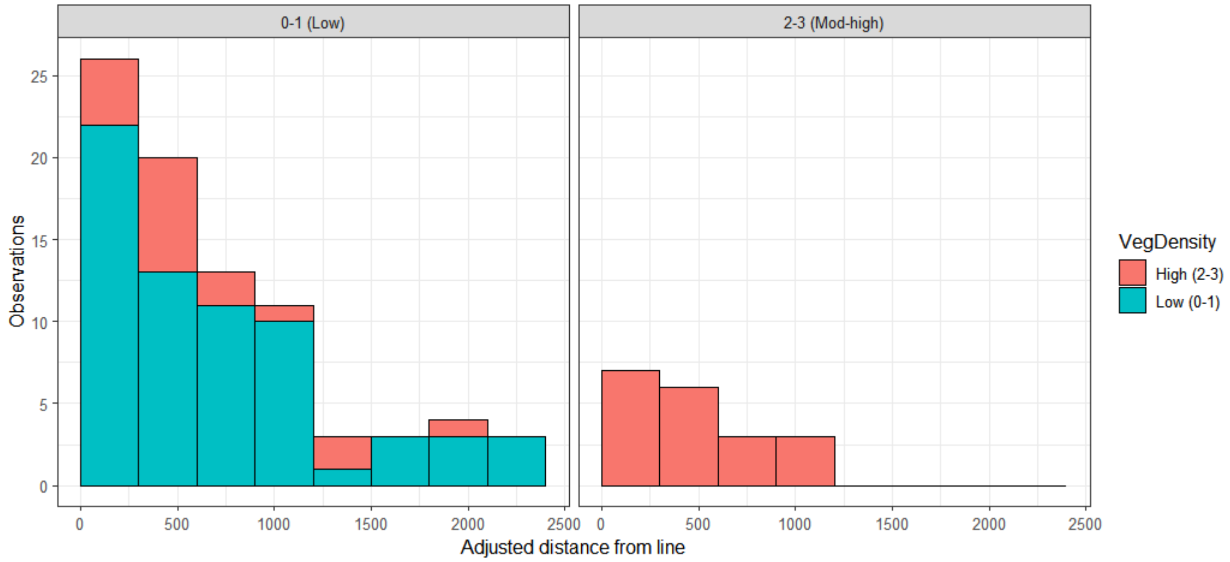

Figure 11: Detection of vegetation height and density (sub-bars).

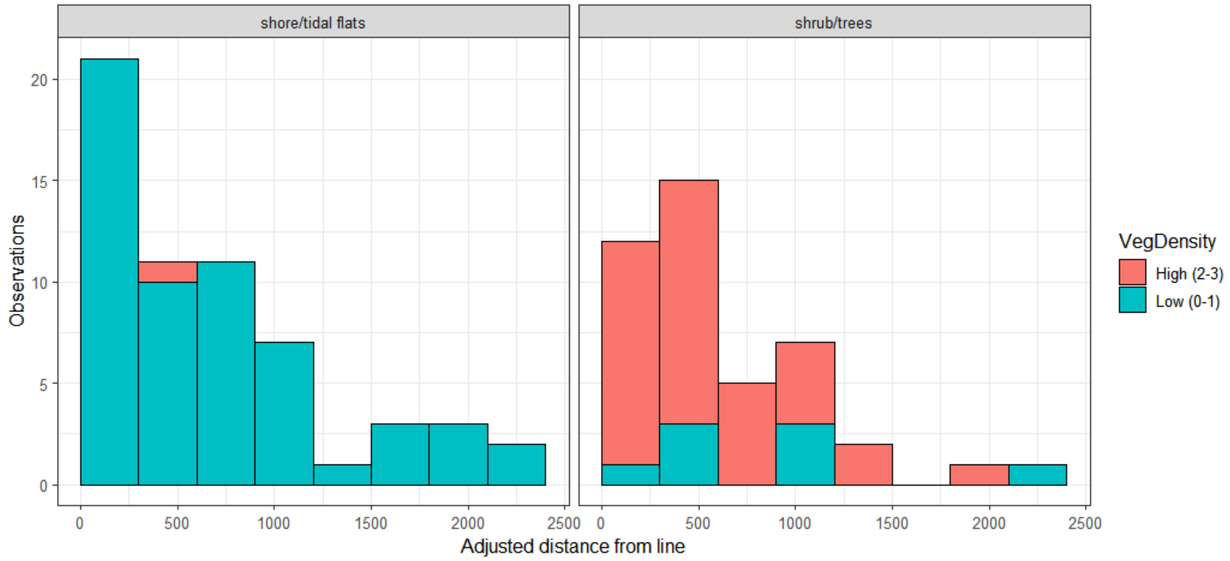

Figure 12: Detection of vegetation height and density (sub-bars).

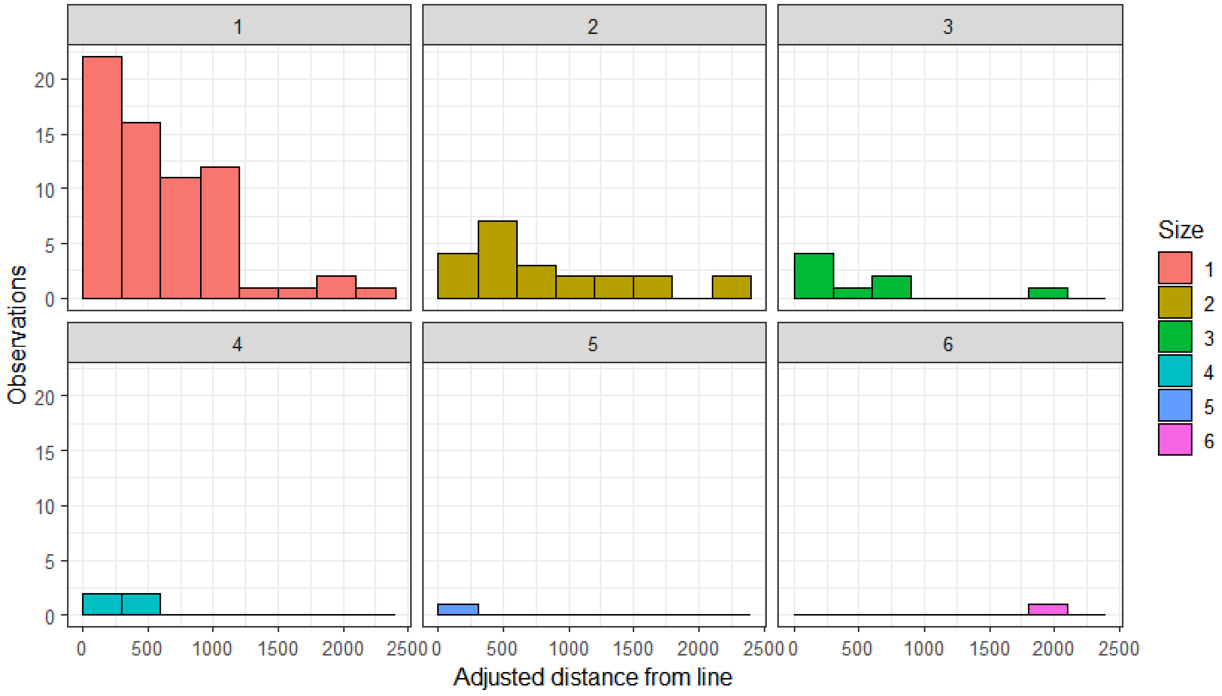

Figure 13: Detection histograms as a function of group size observed.

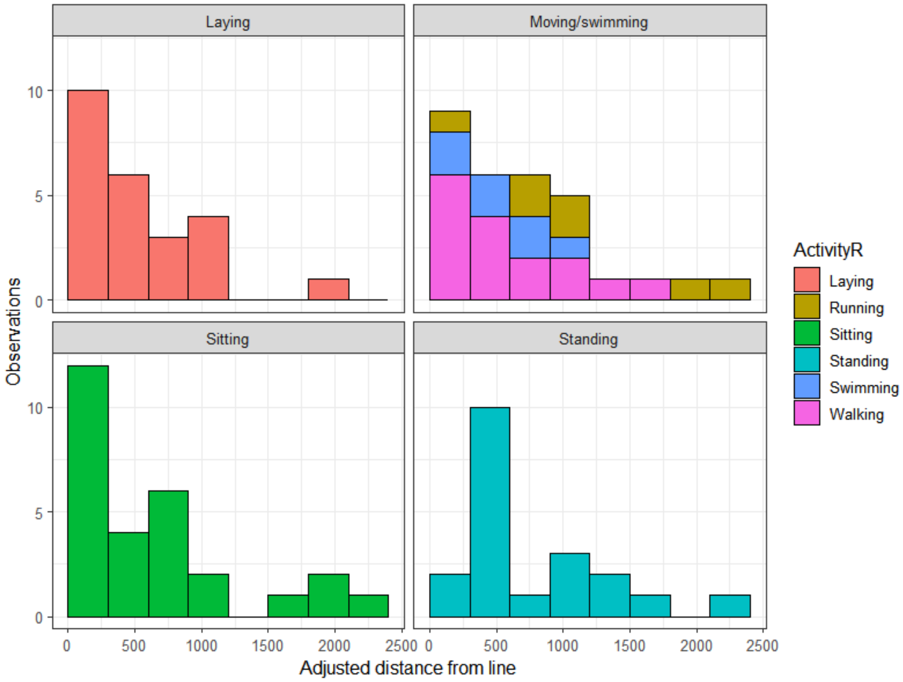

Figure 14: Detection histograms by activity type.

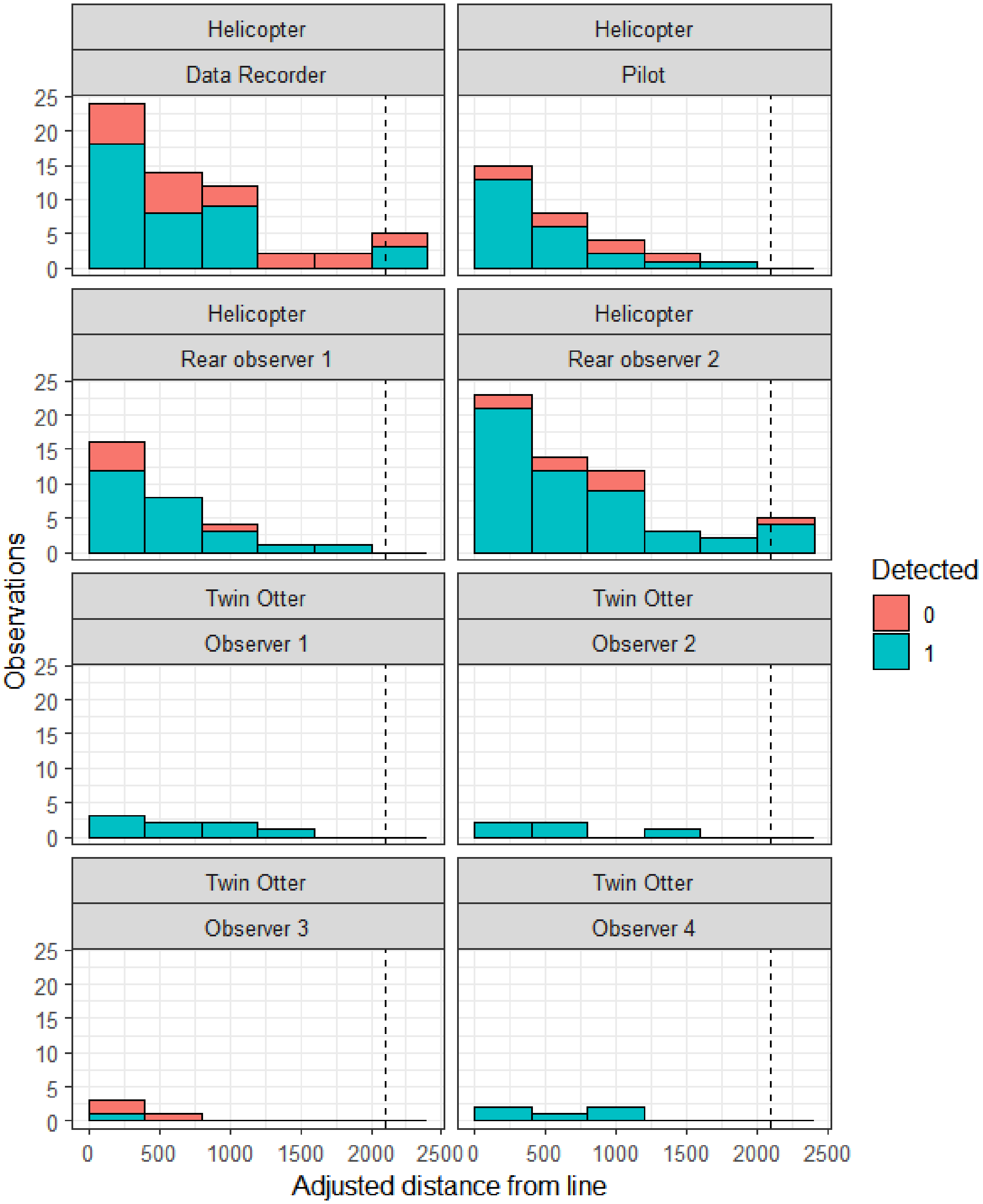

Figure 15: Observer detection histograms for the observers with detections noted (0-not detected, 1-detected)

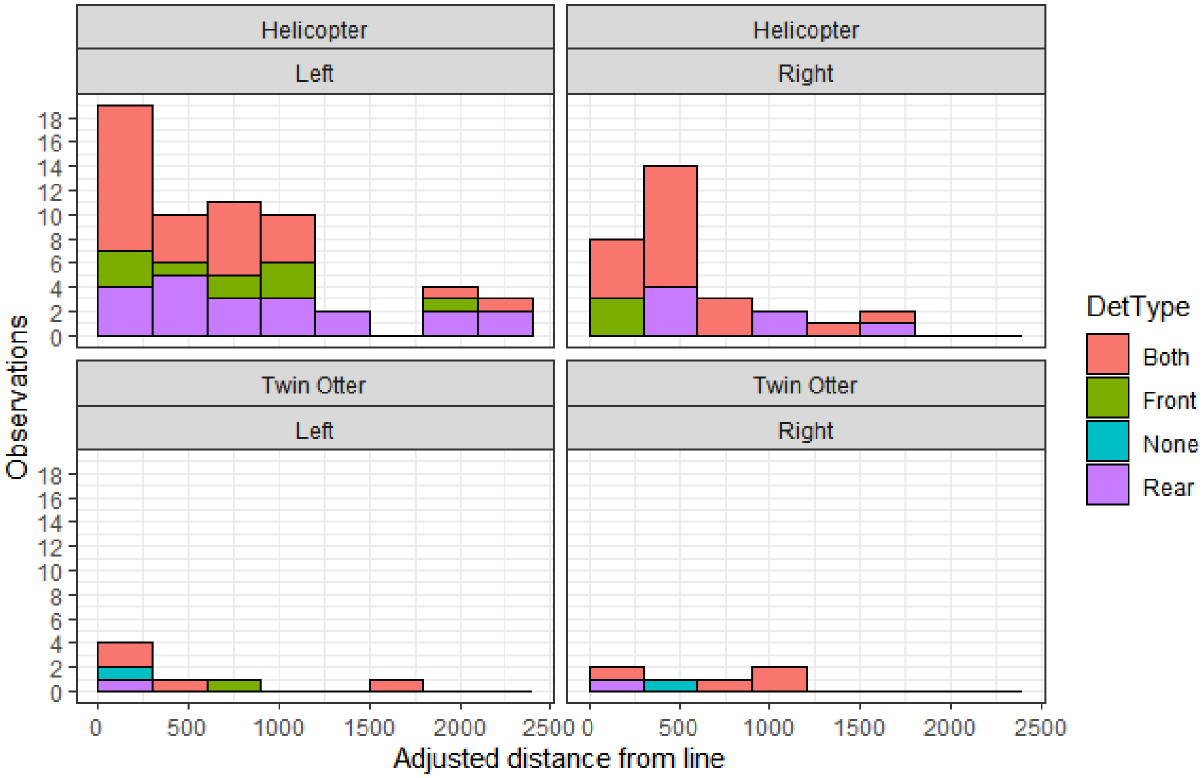

Figure 16: Observations by side of aircraft with detection type noted.

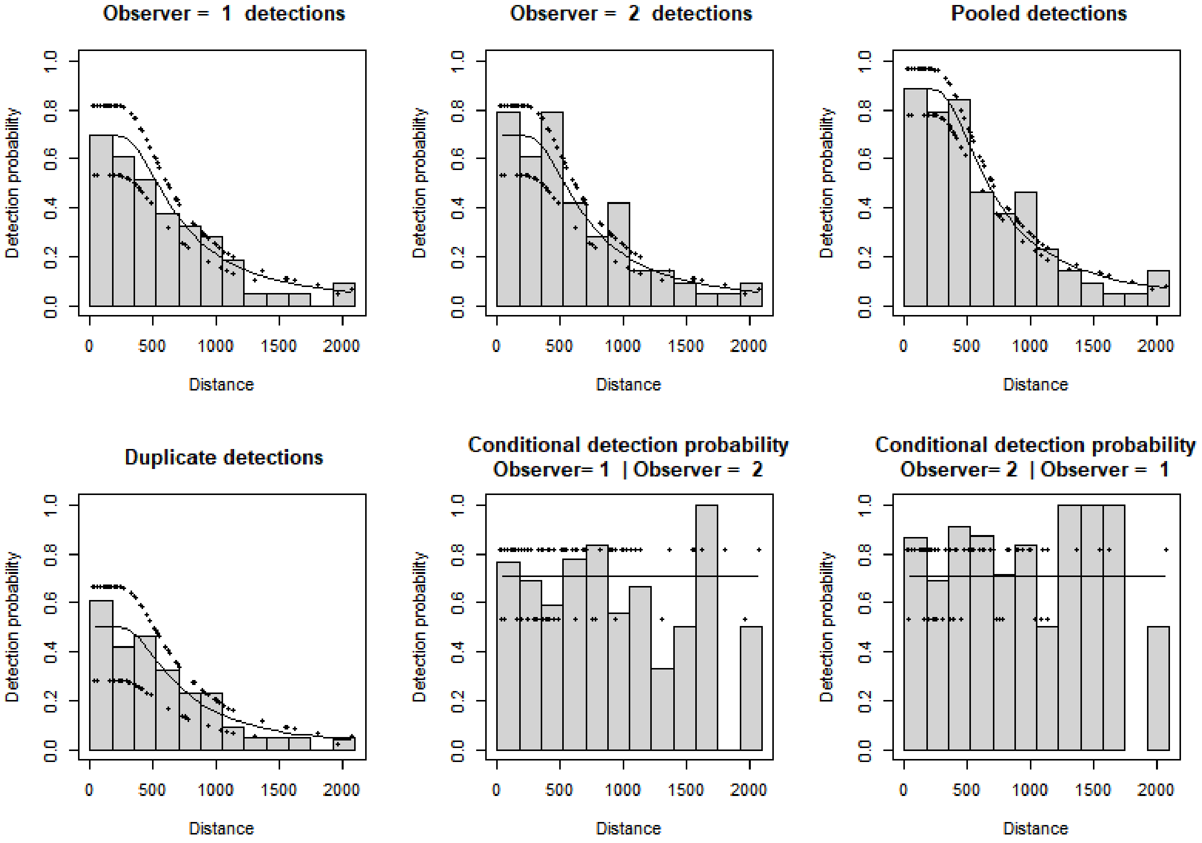

Figure 17: Fit of the most supported distance sampling/double observer model (DS HR(.), MR(VegDensity)) by observers (front and rear). Predictions are given as points for 2 levels of VegDensity.

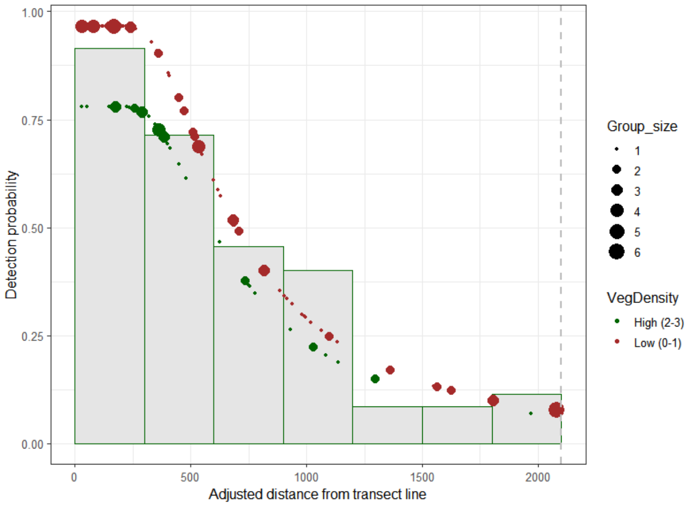

Figure 182: Fit of the most supported distance sampling/double observer model (DS HR(.), MR(VegDensity)) with group sizes and Veg Density delineated.

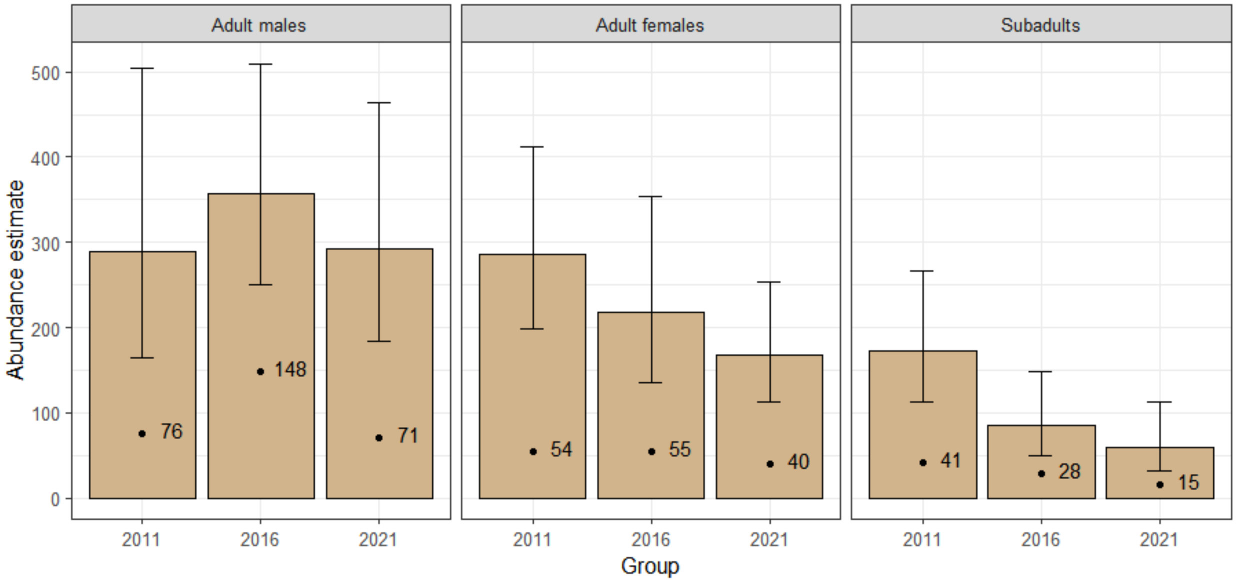

Figure 3: Post-stratified estimates of each age and sex group for 2011, 2016 and 2021. The number of bears observed on transect is also given as a data point.

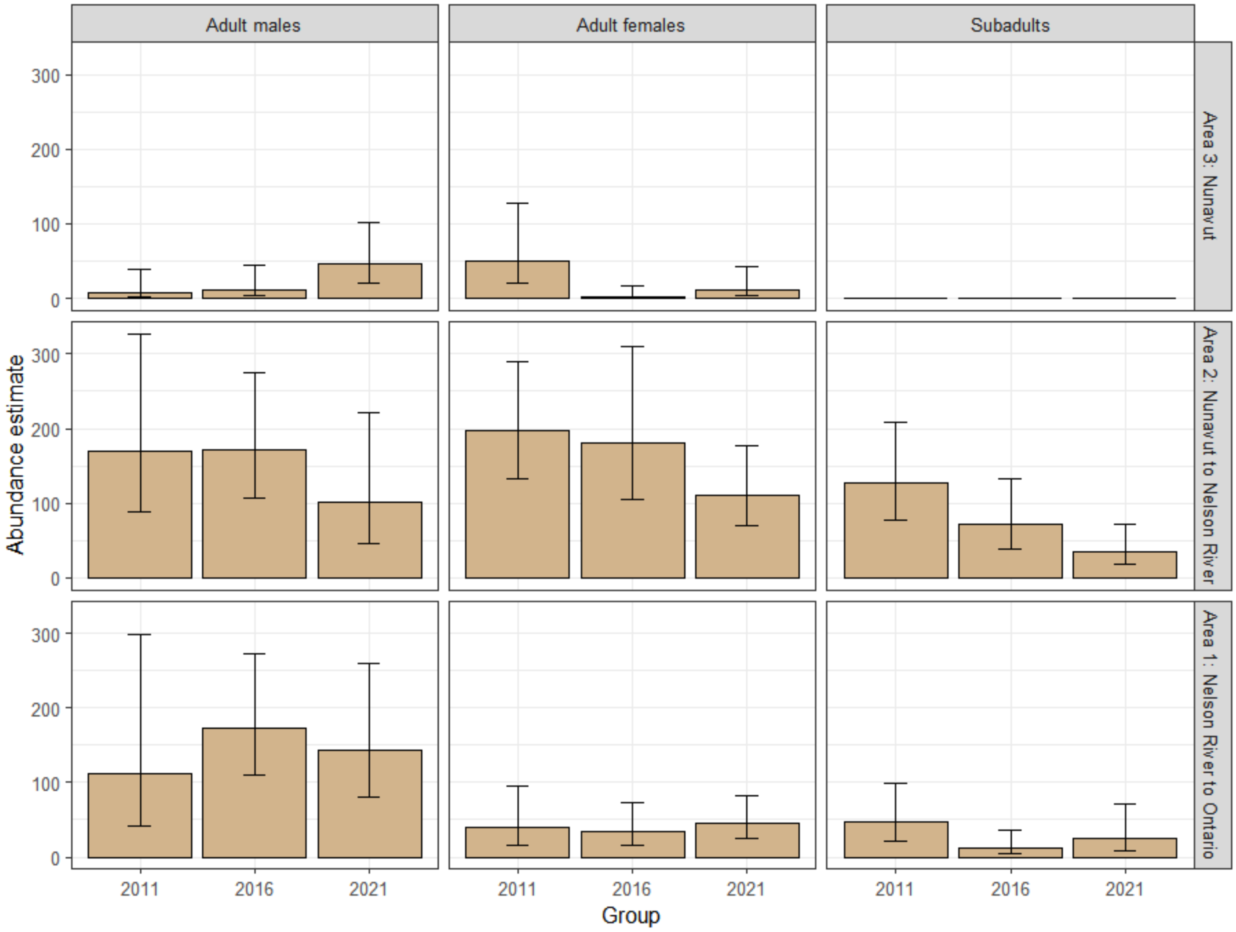

Figure 4: Estimate of abundance by geographic region for adult males, adult female, and subadults.

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Figure 21. Two of three polar bear carcasses encountered during the 2021 WH survey.
(Photo credit: S. Atkinson)

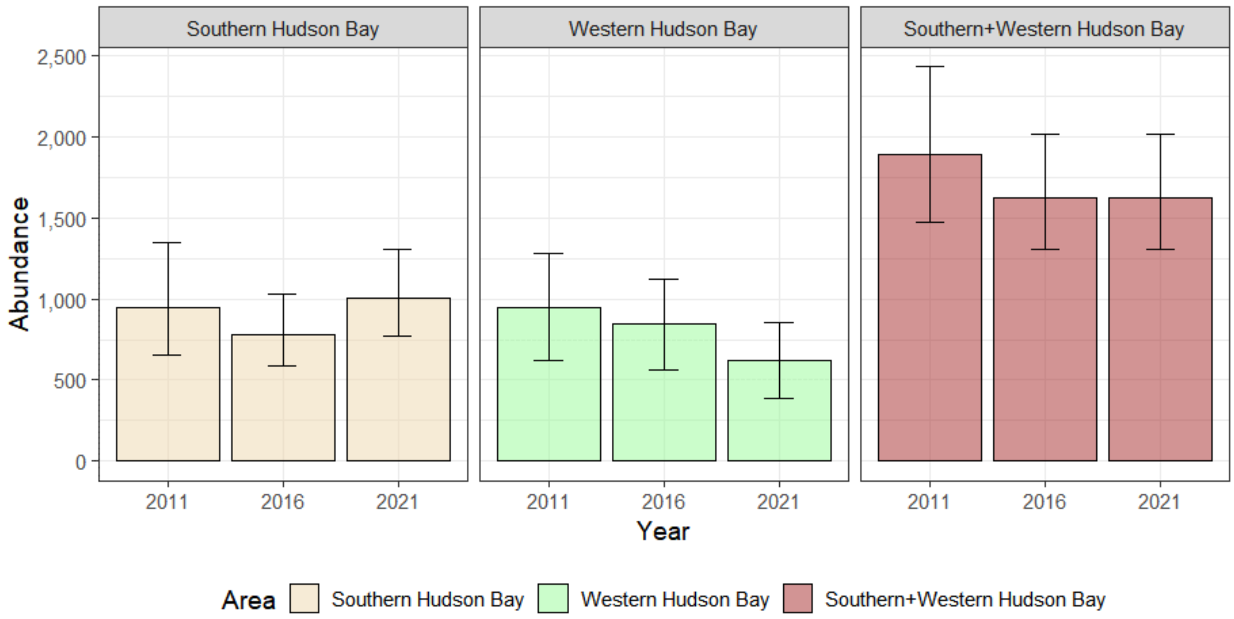

Figure 22. Individual and pooled aerial survey abundance estimates for the Western Hudson Bay (WH) and Southern Hudson Bay (SH), polar bear subpopulations.

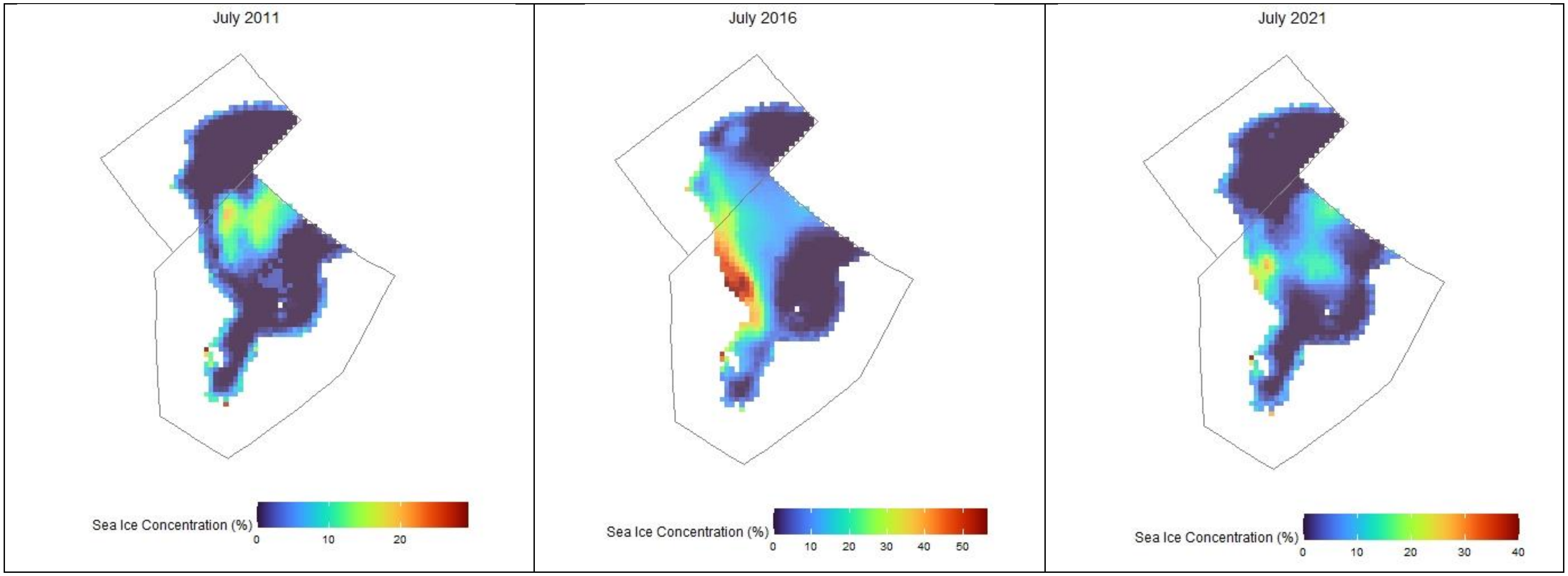

Figure 23. Mean daily sea ice concentration in WH and SH for the month of July from 2011-2021. We used sea ice concentration data from the Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave data set available from the National Snow and Ice Data Center (NSIDC). The monthly average was calculated for each 25 x 25 km grid cell over the eleven-year period.

Table 1. Description of survey strata used in the 2016 and 2021 WH polar bear aerial surveys. (Source: Dyck et al. 2017).

Strata Name	Description
Very Low Density	These strata and transects represented the inland portions of the survey area outside of the Wapusk National Park high density stratum boundaries (Figure 2). These strata were divided further into two main areas, one north and west of the Churchill River up to the Nunavut/Manitoba boundary in the north, and the second south and east of the Nelson River bounded to the east by Cape Tatnam. The very low-density strata covered only inland transects generally ending within 20 to 30 km of the Hudson Bay coastline. Transect spacing was irregular but averaged 17 km across the strata.
Low Density	The stratum and transects occupied the northern extents of the WH polar bear population boundary (approximately 20 km south of Chesterfield Inlet) to the Nunavut/Manitoba border (Figure 2). Modifications from Stapleton et al. (2014) included IQ-based transect extensions both over water and inland within the northern extent of this stratum. Overwater extensions within the remaining extents including 2 transects bi-secting Sentry Island were derived solely from Inuit Qaujimajatuqangit (IQ) reports and recommendations. Transect lines in this stratum were spaced 10 km apart, and extended up to 90 km inland, and up to 30 km into Hudson Bay beyond the coast to incorporate the many offshore islands characterizing this coastline. The development of this stratum was largely based on local knowledge which strongly recommended the extension of coastal transects inland and across open water and coastal islands.
Moderate Density	These strata and transects were divided into two areas, one north and west of the Churchill River up to the Nunavut/Manitoba boundary in the north, and the second south and east of the Nelson River, approximately 60 km east into Ontario to the eastern extent of the WH polar bear population boundary. These strata primarily covered a Hudson Bay coastal strip that was approximately 20 to 30 km wide. Transect spacing within this stratum was 7 km with transects extended beyond the tidal flats into open water. Recent information collected by the Manitoba Department of Sustainable Development on summer and spring polar bear habitat including denning sites, spring emergence habitat, and coastal summer retreat, led this survey effort to modify Stapleton et al. (2014) survey design to define a moderate-density stratum from Cape Tatnam east toward East Penn Island with transects extending beyond the coastal strip up to 70 km inland into known denning habitat (Figure 2).
High Density	The stratum and transects followed those described by Stapleton et al. (2014). The stratum boundary ran between the Churchill River in the west to the coast of Hudson Bay in the east with Churchill forming the northern boundary and the Nelson River approximating the southern boundary. The core of the high-density stratum included Wapusk National Park which is known to be a high density summering area, and further inland, a heavily used denning area (Lunn et al. 2016). Transects in this stratum extended up to 100 km inland and were spaced 6 km apart. As with all other survey strata, all transects were extended 5-30 km beyond the coast into Hudson Bay which enabled the survey design to include bears either in water or on the extensive tidal flats known to be occupied by bears during summer and fall periods (Dyck, 2001; Clark and Stirling 1997).

Table 2. Covariates collected for each polar bear observation.

Covariate	Description
Activity when sighted	Sitting, lying, running, walking, swimming, other
Vegetation Height	1 = <1m, 2 = 1-3m, 3 = >3m
Vegetation Density	1 = sparse/tundra, 2 = moderate, 3 = dense
Habitat Class	1 = open, 2 = water, 3 =shore/tidal flats, 4 = shrub (below bear head height), 5 = shrub (above bear head height) or trees
Habitat Description	Specify general habitat type (e.g. Coastal plain, tundra, beach, rocky coast, island)
Visibility	1= poor, 2 = fair, 3 = excellent
Cloud Cover	Clear, broken, overcast
Glare	Effect on observer ability to see bears. 1 = no effect, 2 = moderate, 3 = strong.

Table 3: Covariates considered in the mark-recapture/distance sampling analysis. The primary use of the covariate for distance sampling analysis (DS) and mark-recapture analysis (MR) is denoted.

Covariate	Type	DS	MR	description
size	continuous	x	x	group size
aircraft	binary	x	x	helicopter or airplane
helip	binary	x	x	Pilot of helicopter
helir	binary	x	x	Recorder/Navigator of helicopter.
hab	categorical	x	x	habitat within 30m of observation as classified by observers (Open, Water, Shore, low shrub, tall shrub, and Tree)
RSveg	categorical	x	x	Landsat habitat (Gravel, Low vegetation, Shrub, Tree, and water) at pixel (625 m^2) scale
Veg_height	continuous	x	x	Relative height of vegetation (0-3)
Veg_density	continuous	x	x	Relative density of vegetation (0-3)
vis	binary	x	x	Visibility based on weather
observer	categorical		x	Observers (12)
side	categorical	x	x	Side of plane
Heliside	Categorical	x	x	Fixed-wing, heli right and heli left categories
glare	continuous	x	x	Sun altitude; only in equation if sun was facing observer
Activity	categorical	x	x	Activity of bear when first observed

Table 4. Summary of bears counted on and off transect during distance sampling for the Western Hudson Bay survey 2021.

Strata	On/off transect	Bears observed	Groups observed	Mean group size	SD group size	min	max
High	On	92	60	1.53	0.87	1	5
Low	On	14	9	1.56	1.01	1	4
Moderate N	On	9	5	1.80	0.84	1	3
Moderate S	On	61	38	1.61	1.14	1	6
Very Low S	On	1	1	1.00	0.00	1	1
total		176	112	1.57	0.97	1	6
High	Off	10	6	1.67	1.63	1	5
Low	Off	3	3	1.00	0.00	1	1
Moderate N	Off	2	2	1.00	0.00	1	1
Moderate S	Off	3	2	1.50	0.71	1	2
Total		18	13	1.38	1.12	1	5
Total (on+off)		194	125	1.55	0.98	1	6

Table 5: Summary of bears included and excluded from the distance analysis based on left truncation (measured blind spot for twin otter=99 m, ASTAR-73.5m on each side of aircraft), right truncated (2100 meters after left truncation subtracted). Also, 2 bears were only observed by data recorders and were not included in the analysis.

Strata	**Left Truncated**	**Included in analysis**	**Only observed by data recorder**	**Right truncated**
High	13	76	0	3
Low	0	13	1	0
Moderate N	1	7	1	0
Moderate S	1	58	0	2
Total	15	154	2	5

Table 6: Summary of observer detection frequencies and naïve detection probabilities based on frequencies of detections. Note that sample sizes pertain to an event (bears seen on a side of the aircraft) and therefore will be double the actual number of observations (given that 2 observers were involved in each observation).

Aircraft	**Observer**	**Observed**	**missed**	**total**	**naïve p**
Heli	pilot	23	7	30	0.77
Heli	Navigator/Data Recorder	38	21	59	0.64
Heli	1	25	5	30	0.83
Heli	2	51	8	59	0.86
Otter	1	7	1	8	0.88
Otter	2	5	1	6	0.83
Otter	3	5	1	6	0.83
Otter	4	1	4	5	0.20
Otter	5	1	0	1	Constant

Table 7: Model selection results for distance sampling analysis. The mark-recapture component of the MRDS model was set at constant for this analysis step. Covariates are listed in Table 1. The detection function (hr=hazard rate, hn=half normal) is shown along with covariates. Constant models are shaded. Akaike information criterion (AIC), the differences between AIC of the given model and most supported model ∆AIC, Akaike weight (w_i), and Log-likelihood of each model is also shown.

No	DF	Detection function model	AICc	∆AICc	w_i	K	LogL
1	hr	constant	1621.73	0.00	0.13	3	-807.7
2	hr	VegDensity	1622.14	0.41	0.10	4	-806.9
4	hr	Helirecorder	1622.72	0.98	0.08	4	-807.1
5	hr	HabClassP	1622.91	1.17	0.07	4	-807.2
6	hr	Side	1623.11	1.38	0.06	4	-807.3
7	hr	VegDensity + Helirecorder	1623.15	1.41	0.06	5	-806.3
8	hn	constant	1623.19	1.45	0.06	2	-809.5
10	hn	size	1623.60	1.86	0.05	3	-808.7
11	hr	VegDensity + Side	1623.68	1.95	0.05	5	-806.5
12	hr	Glare	1623.70	1.96	0.05	4	-807.6
13	hr	size	1623.71	1.97	0.05	4	-807.6
14	hr	Aircraft	1623.90	2.17	0.04	4	-807.7
15	hr	VegDensity + VegHeight	1624.06	2.32	0.04	5	-806.7
16	hr	VegDensity + HabClassP	1624.36	2.63	0.03	5	-806.9
17	hr	HeliSide	1624.60	2.87	0.03	5	-807.0
18	hr	RSveg3	1625.02	3.29	0.02	5	-807.2
19	hr	VegDensity + HabClassP + Helir	1625.38	3.65	0.02	6	-806.2
20	hr	Strata	1626.05	4.32	0.01	6	-806.6
	hr	ObName	1626.51	4.77	0.01	8	-804.5
21	hr	ActivityP	1627.04	5.31	0.01	6	-807.1
	hr	HabClassType	1628.01	6.28	0.01	7	-806.4

Table 8: Model selection results for double observer analysis. The most supported distance sampling model (HR constant) was used for the distance sampling component. Covariates are listed in Table 1. The detection function (hr=hazard rate, hn=half normal) is shown along with covariates. Constant models are shaded. Akaike information criterion (AIC), the differences between AIC of the given model and most supported model ∆AIC, Akaike weight (*w_i*), and Log-likelihood of each model is also shown.

No	Double observer model	AICc	∆AICc	w_i	K	LogL
1	VegDensity	1613.95	0.00	0.30	4	-802.8
2	VegDensity + HeliSide	1615.05	1.10	0.17	6	-801.1
3	VegHeight + VegDensity	1615.81	1.86	0.12	5	-802.6
4	HabClassP	1615.89	1.95	0.11	4	-803.7
5	RSveg3	1616.84	2.89	0.07	5	-803.1
6	VegHeight	1617.82	3.88	0.04	4	-804.7
7	Heli recorder	1617.99	4.04	0.04	4	-804.8
8	Heli pilot	1618.41	4.47	0.03	4	-805.0
9	Position	1619.15	5.20	0.02	4	-805.4
10	Activity	1619.27	5.33	0.02	6	-803.2
11	Helip + Helir	1619.56	5.62	0.02	5	-804.5
12	Side	1620.71	6.76	0.01	4	-806.1
13	ObName	1621.09	7.14	0.01	11	-798.0
14	size	1621.22	7.27	0.01	4	-806.4
15	Constant	1621.73	7.79	0.01	3	-807.7
16	Glare	1621.98	8.03	0.01	4	-806.8
17	HeliRight+HeliLeft+Fixedwing	1622.04	8.09	0.01	5	-805.7
18	HeliRight (only)	1622.27	8.33	0.00	6	-804.7
19	AirPosition	1622.55	8.60	0.00	4	-807.1
20	Aircraft	1623.85	9.91	0.00	7	-804.3
21	Helip + Helir +jb +vt	1624.06	10.11	0.00	6	-805.6
22	Strata	1624.35	10.40	0.00	6	-805.7

Table 9: Model selection results for composite distance sampling and double observer analysis. The most supported distance sampling model (HR constant) was used for the distance sampling component. Covariates are listed in Table 1. The detection function (hr=hazard rate, hn=half normal) is shown along with covariates. Constant models are shaded. Akaike information criterion (AIC), the differences between AIC of the given model and most supported model ∆AIC, Akaike weight (w_i), and Log-likelihood of each model is also shown. Abundance estimates across all strata are given for reference.

Model selection									**Abundance**	
No	**DSDF**	**DS model**	**Double ob model**	**AICc**	**∆AICc**	**w_i**	**K**	**LogL**	**N**	**CV**
1	hr	constant	VegDensity	1613.94	0.00	0.13	4	-802.8	639	19.2%
2	hr	VegDensity	VegDensity	1614.39	0.45	0.10	5	-801.9	608	18.6%
3	hr	Hellir	VegDensity	1614.96	1.02	0.08	5	-802.2	619	19.2%
4	hr	constant	VegDensity + HeliSide	1615.03	1.09	0.07	6	-801.1	648	19.2%
5	hr	HabClassP	VegDensity	1615.15	1.21	0.07	5	-802.3	605	18.5%
6	hn	constant	VegDensity	1615.35	1.41	0.06	3	-804.6	578	15.7%
7	hr	VegDensity + Bellr	VegDensity	1615.44	1.50	0.06	6	-801.3	604	18.8%
8	hr	VegDensity	VegDensity + HeliSide	1615.57	1.64	0.06	7	-800.2	619	18.8%
9	hr	constant	VegDensity + Ob_jb	1615.63	1.69	0.05	5	-802.5	638	19.1%
10	hr	constant	Veg_Height+ VegDensity	1615.79	1.85	0.05	5	-802.6	639	19.1%
11	hr	HeliRight	VegDensity + HeliSide	1615.87	1.93	0.05	7	-800.3	598	18.7%
12	hr	size	VegDensity	1615.95	2.01	0.05	5	-802.7	671	21.8%
13	hr	VegDensity + Side	VegDensity	1615.97	2.03	0.05	6	-801.5	603	18.8%
14	hr	HeliSide	VegDensity + HeliRight	1616.27	2.33	0.04	7	-800.5	599	18.8%
15	hr	VegDensity	HabClass	1616.34	2.40	0.04	5	-802.9	596	18.3%
16	hr	Veg_HeightP+ VegDensity	VegDensity	1616.34	2.41	0.04	6	-801.7	607	18.5%
17	hr	VegDensity+ HeliSide	VegDensity	1617.69	3.75	0.02	7	-801.2	599	18.7%
18	hr	constant	constant	1621.73	7.79	0.00	3	-807.7	622	19.1%
19	hr	VegDensity + HeliSide	HeliSide	1625.92	11.98	0.00	8	-804.2	578	18.1%

Table 10: Strata estimates from the most supported DS/MR model (Table 9, model 1)

Strata	individuals	N	SE	Conf. Limit		CV
High	76	290	75.01	175	482	0.26
Low	13	76	32.77	33	174	0.43
Moderate N	7	28	21.39	7	113	0.76
Moderate S	58	244	57.08	154	387	0.23

Table 11. Post-stratified estimates of sex and age groups from model 1 (Table 9).

Group	Strata	Individuals	N	SE	CIL	CIU	CV
Males	High	25	89.3	39.7	38	210	0.44
Males	Low	8	46.5	18.9	21	102	0.41
Males	Moderate N	3	12.1	8.9	3	47	0.74
Males	Moderate S	35	144.0	43.1	80	259	0.30
Males	Very Low S	0	0.0	0.0	0	0	0.00
Males	Total	71	292.0	69.2	184	463	0.24
Females	High	26	102.7	24.5	64	164	0.24
Females	Low	2	11.6	8.5	3	43	0.73
Females	Moderate N	2	8.0	8.0	1	45	1.00
Females	Moderate S	10	46.1	13.9	26	83	0.30
Females	Very Low S	0	0.0	0.0	0	0	0.00
Females	Total	40	168.5	35.2	112	253	0.21
Cubs	High	9	35.9	15.3	16	82	0.43
Cubs	Low	1	5.8	5.9	1	31	1.01
Cubs	Moderate N	1	4.0	4.0	1	22	1.00
Cubs	Moderate S	1	4.9	4.9	1	27	1.01
Cubs	Very Low S	0	0.0	0.0	0	0	0.00
Cubs	Total	12	50.6	18.2	25	101	0.36
Yearlings	High	7	27.4	11.6	12	62	0.42
Yearlings	Low	2	11.6	12.0	2	64	1.03
Yearlings	Moderate N	1	4.0	4.0	1	22	1.00
Yearlings	Moderate S	6	24.7	13.2	9	68	0.53
Yearlings	Very Low S	0	0.0	0.0	0	0	0.00
Yearlings	Total	16	67.7	22.8	35	129	0.34
Subadults	High	9	35.1	12.7	17	71	0.36
Subadults	Low	0	0.0	0.0	0	0	0.00
Subadults	Moderate N	0	0.0	0.0	0	0	0.00
Subadults	Moderate S	6	24.7	13.7	9	70	0.56
Subadults	Very Low S	0	0.0	0.0	0	0	0.00
Subadults	Total	15	59.8	19.5	32	112	0.33

Table12. Estimates of gross change (GC), probability that gross change is greater than 1 p(GC<1), and annual rate of change (λ) at different survey intervals for distance sampling estimates.

Interval Group	GC	SE	Conf. Limit		p(GC>1)	λ	SE	Conf. limit	
2011 to 2016									
Males	1.24	0.46	0.66	2.43	0.759	1.04	0.07	0.92	1.19
Females	0.76	0.25	0.41	1.37	0.177	0.95	0.06	0.84	1.06
Subadults	0.49	0.19	0.24	0.97	**0.021**	0.87	0.06	0.75	0.99
All	0.87	0.22	0.54	1.39	0.282	0.97	0.05	0.89	1.07
2016 to 2021									
Males	0.82	0.26	0.45	1.44	0.237	0.96	0.06	0.85	1.08
Females	0.77	0.27	0.42	1.45	0.214	0.95	0.06	0.84	1.08
Subadults	0.70	0.34	0.30	1.59	0.193	0.93	0.08	0.79	1.10
All	0.77	0.20	0.47	1.25	0.144	0.95	0.05	0.86	1.05
2011 to 2021									
Males	1.01	0.41	0.50	2.10	0.525	1.00	0.04	0.93	1.08
Females	0.59	0.17	0.34	1.01	**0.027**	0.95	0.03	0.90	1.00
Subadults	0.35	0.15	0.16	0.72	**0.002**	0.90	0.03	0.83	0.97
All	0.67	0.18	0.40	1.10	**0.056**	0.96	0.02	0.91	1.01

Table 13. Estimates of gross change (GC), probability that gross change is greater than 1 and annual rate of change (λ) from 2011 to 2021 for distance sampling estimates for age/sex groups by geographic region. Estimates are given for adult females, adult males and subadults.

Area Group	GC	SE	Conf. Limit		p(GC>1)	λ	SE	Conf. Limit	
Area 3: Nunavut									
Males	6.60	16.34	1.40	55.55	0.990	1.21	0.12	1.03	1.49
Females	0.23	0.27	0.04	1.00	**0.025**	0.86	0.07	0.73	1.00
Subadults		None observed							
Area 2 Nunavut to Nelson River									
Males	0.60	0.36	0.22	1.58	0.145	0.95	0.05	0.86	1.05
Females	0.56	0.18	0.31	1.02	**0.028**	0.94	0.03	0.89	1.00
Subadults	0.28	0.13	0.12	0.63	**0.001**	0.88	0.04	0.81	0.95
Area 1: Nelson River to Ontario									
Males	1.29	1.00	0.46	4.21	0.720	1.03	0.06	0.93	1.15
Females	1.18	0.81	0.44	3.49	0.661	1.02	0.05	0.92	1.13
Subadults	0.53	0.44	0.14	1.76	0.140	0.94	0.06	0.82	1.06

Table 14. Polar bear litter sizes and number of dependent offspring observed (as proportion of total observations) during recent ice-free season studies in central and eastern Canada. Data are presented as mean (standard error).

Subpopulation	Litter Size		Proportion of Total Observations		Source
	Cubs-of-the-year	Yearlings	Cubs-of-the-year	Yearlings	
Western Hudson Bay (2011)	1.43 (0.08)	1.22 (0.10)	0.07	0.03	Stapleton et al. (2014)
Western Hudson Bay (2016)	1.63 (0.10)	1.25 (0.16)	0.11	0.03	Dyck et al. (2017)
Western Hudson Bay (2021)	1.46 (0.13)	1.39 (0.18)	0.09	0.09	This report
Southern Hudson Bay (2011)	1.56 (0.06)	1.54 (0.08)	0.16	0.12	Obbard et al. (2015)
Southern Hudson Bay (2016)	1.46 (0.06)	1.32 (0.10)	0.19	0.05	Obbard et al. (2018)
Southern Hudson Bay (2021)	1.57	1.47	0.18	0.18	Northrup et al. (2022)
Baffin Bay (2011-13)	1.57 (0.06)	1.51 (0.09)	0.19	0.10	SWG (2016)
Foxe Basin (2009-2010)	1.54 (0.04)	1.48 (0.05)	0.13	0.10	Stapleton et al. (2015)
Davis Strait (2017-2018)	1.42 (0.15)	1.54 (na)	0.12	0.09	Dyck et al. (2022)

Appendix 1 - Sensitivity Analyses to Truncation and Observer Issues

a) Right Truncation

Model 1 was also run with distances up to the maximum distance (2210 included) to test sensitivity of model fit and estimates to right truncation. Model fit was reduced (overall χ^2=24.1, df=16, p=0.0.088) with an overall estimate of 650 bears (SE=128.1, CI=442-966, CV=19.7%). The reduction of model fit suggested that right truncation was justified with minimal overall change in estimates.

b) Left Truncation

Data was left truncated at intervals further from the predefined distances up to 100 meters with minimal change in estimates (Figure A2-1).

c) Sensitivity to Lower Number of Observations on the Right Side of the Helicopter

The right side of the helicopter had the pilot as the front observer and the same individual as second observer for most of the survey. The number of observations from this side was reduced with 30 groups observed compared to 59 to the left side of the helicopter. The difference in detections was modelled using the HeliSide term that allowed for distinct detection function or detection probabilities for the fixed wing (sides pooled), right side and left side of the helicopter. A model that allowed distinct detection probabilities for each side of the helicopter (Table 7, model 4) showed moderate support from the data (ΔAICc=1.1) with an estimate that was 10 bears higher than the most supported model. The main challenge with this analysis is that the actual detection probabilities for the pilot and right observer as indicated by double observer data suggests reasonable detection probabilities (Table 4) with the pilot and right observer showing naïve detection probabilities of 0.77 and 0.83. Therefore, missing of observations is not evident from these probabilities, which could be due to low sample sizes. Detection plots for each observer (Figure 12) also show observations at the full range of distances and therefore there is little evidence of differences in detection functions.

One approach to assess potential bias is to post-stratify the data by side of aircraft. For this the full data set is used to model detection, and then the right and left side observations are used for estimates. Estimates from the 2 sides then add up to the total estimate (Table A2-1). If this is done approximately 40% of the estimate comes from the right side of aircraft and 60% from the left side. Obviously, it would not be expected for the 2 estimates to be equal, however, this gives a sense of the difference in estimates by side of aircraft.

The main potential bias with the right side of the helicopter would be unmodelled heterogeneity of detection probabilities which are exacerbated when detection probabilities become lower. Having 2 weaker observers paired together makes it harder to assess if this is occurring. The amount of data available to model this potential effect is limiting.

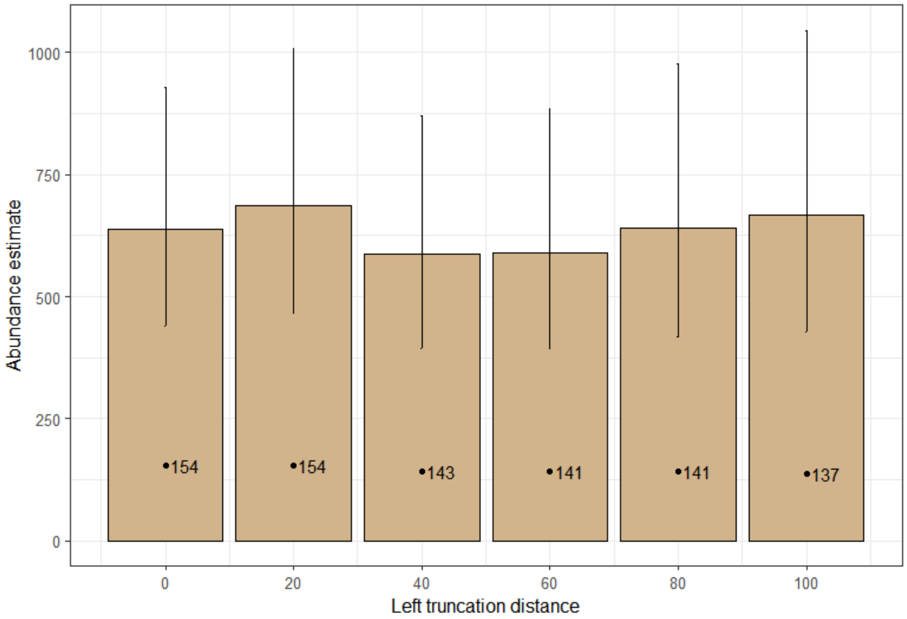

Figure A2-1. Model 1 sensitivity to additional left truncation distances. Points are the sample size of bears used in the analysis.

Table A2-1: Post-stratified estimates by side of aircraft.

Strata	Individuals	N	SE	CIL	CIU	CV
right side						
High	31	115	49.9	50	265	43.4%
Low	9	52	30.2	18	154	57.7%
Moderate N	2	8	8.1	1	45	100.6%
Moderate S	18	79	29.0	38	161	36.9%
Very Low S	0	0	0.0	0	0	0.0%
Total	60	254	71.1	148	437	28.0%
left side						
High	45	175	41.5	110	279	23.7%
Low	4	23	11.8	9	61	50.5%
Moderate N	5	20	20.1	4	112	100.0%
Moderate S	40	166	40.3	103	267	24.3%
Very Low S	0	0	0.0	0	0	0.0%
Total	94	385	73.9	264	560	19.2%
Combined Total (left +right)	154.00	638.56				

Appendix 2 – Further estimation of trend for age-sex post stratified estimates

We used a simulation methodology to estimate confidence limits on gross and annual change in post-stratified estimates of males, adult females (excluding cubs and yearlings), subadults, and all bears. This simulation test is equivalent to a one-sided hypothesis test for decline where the null hypothesis is that the estimate in the previous year is equal or greater than the current year H_o: $N_2 \geq N_1$ and the alternative hypothesis is that the current year estimate is lower (Ha: $N_2 < N_1$). We cross-validated these results using a standard t-test (Satterthwaite 1946, Zar 1996). Comparison of p-values and pGC reveals that these two test methods yielded very similar results.

We note that the 1-tailed hypothesis provides a more powerful test for decline than a 2-tailed test (which tests if the 2 estimates are equal). We felt the 1 tailed test was justified given that the question of management interest was whether a decline was occurring as opposed to whether the two estimates were equal. We also note that overlap of confidence intervals (Figure 20) is not a valid test for statistical significance especially if the hypothesis being tested is one tailed.

Table 1: Results of t-tests for decline in yearly estimates of age/sex groups for WHB (null hypothesis H_o: $N2 \geq N_1$ and alternative hypothesis H_a: $N_2 < N_1$). Degrees of freedom (df) for each survey and combined degrees of freedom are also given.

Year			Estimates					t-test		simulation
group	N_1	SE_N_1	df_N_2	N_2	SE_N_2	df_N_2	t-test	df (N_1&N_2)	p-value (1-tailed)	p(GC<1)
<u>2011-6</u>										
Males	289	82.9	91.4	357	65.0	224.6	0.65	206.4	0.741	0.759
Females	286	53.1	204.5	219	54.0	154.3	-0.89	350.1	0.186	0.177
Subadults	173	38.3	124.0	85	24.2	75.3	-1.94	192.4	0.027	0.021
All	956	166.3	244.9	831	138.6	217.5	-0.58	455.7	0.282	0.281
<u>2016-21</u>										
Males	357	65.0	224.6	292	69.2	170.0	-0.68	378.9	0.248	0.236
Females	219	54.0	154.3	168	35.2	192.8	-0.78	273.9	0.219	0.216
Subadults	85	24.2	75.3	60	19.5	108.9	-0.82	158.7	0.207	0.194
All	831	138.6	217.5	639	122.3	198.9	-1.04	413.7	0.149	0.144

<u>2011-21</u>										
Males	289	82.9	91.4	292	69.2	170.0	0.03	208.7	0.512	0.525
Females	286	53.1	204.5	168	35.2	192.8	-1.85	351.6	0.033	0.027
Subadults	173	38.3	124.0	60	19.5	108.9	-2.63	182.8	0.005	0.002
All	956	166.3	244.9	639	122.3	198.9	-1.54	427.5	0.062	0.056

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- ᐊᑐᖅᓯᖃᑦᑕᖅᑐᒍᑦ ᓂᕿᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕕᓂᖏᑦ ᐱᔭᐅᒃᑲᓐᓂᕈᑎᑦ ᓇᓄᐃᑦ ᖃᐅᔨᒪᔾᔪᑎᒃᓴᑦ ᖃᐅᔨᓂᐊᕐᓗᒋᑦ ᓄᒃᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐊᕙᑖᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ

- 727 ᐊᐅᖏᑕ ᐃᓄᖏᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓚᐅᖅᑐᑦ ᐱᔭᐅᓪᓗᑎᒃ ᐅᖓᑖᓂᑦ 6 ᐊᕐᕌᒍᓄᑦ, ᑖᒃᑯᓇᖓᑦ ᐱᑕᖃᖅᑐᖅ 458 ᐊᔾᔨᒌᓐᖏᑦᑐᑦ ᐊᑐᓂ ᓇᓗᓇᕈᓐᓃᖅᑎᑦᑎᓚᐅᖅᑐᖅ

- ᑕᐃᑲᓂ 2021 ᐊᒻᒪᓗ 2022, 25-36% ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᓇᓄᐃᑦ ᓇᓂᔭᐅᓚᐅᖅᑐᑦ ᓄᒃᑕᖅᑐᑦ ᐊᕙᑖᓂᑦ ᒪᕐᕉᖕᓂᑦ ᐃᓚᖏᓐᓂᑦ ᓇᓗᓇᕈᓐᓃᓚᐅᖅᑐᖅ ᐊᖏᔪᒻᒪᕆᖕᒥᒃ ᓄᒃᑕᓲᑦ ᐊᕙᑖᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᑭᒡᓕᖏᓐᓂ.

- ᐃᓗᐊᓂ ᐊᕐᕌᒍᒥ ᐊᔾᔨᒌᖃᑦᑕᓐᖏᑐᑦ ᓇᒧᖔᐅᕙᒍᓯᖏᑦ ᑕᕆᐅᖅ ᓯᑯᐊᓂᒃ ᒪᓂᒃᑐᑦ ᓇᒧᖐᓯᒪᓂᖏᓐᓂᒃ ᓇᓄᐃᑦ.

- ᓇᓄᐃᑦ ᓄᒃᑕᕐᓂᖏᑦ ᐊᕙᑖᓂᑦ ᖃᓂᒌᑦ ᓇᓅᖃᑎᖏᑦ ᐅᓄᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐅᖃᐅᓯᐅᓚᐅᖅᑐᓂᑦ ᑭᓯᐊᓂ ᑕᐃᒪᓐᓇᐅᓚᐅᖅᑐᑦ ᓂᒋᖅᐸᓯᐊᓂ ᑭᒡᓕᐊᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (Cape Tatnam) ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ. ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᓇᓄᐃᑦ ᓄᒃᑕᕐᓂᖏᑦ ᐊᒻᒪᓗ ᓇᓐᓄᒃᑕᐅᖃᑦᑕᖅᑐᑦ ᑕᒪᐃᓐᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᒥ ᑭᓯᐊᓂᐅᖃᑦᑕᖅᑐᑦ ᓇᓐᓄᒃᑕᐅᓂᐊᖅᑐᑦ ᐊᐅᓚᑕᐅᓪᓗᑎᒃ ᑐᑭᓕᐅᕈᑕᐅᔪᑦ ᐋᕿᐅᑎᑦᑎᐊᕐᓂᐊᕐᒪᑕ.

ᖃᓄᐃᓕᐅᕆᐊᖃᖅᑐᓄᑦ ᓇᐃᓈᖅᓯᒪᔪᖅ

ᐊᖑᓇᓱᒃᑕᓄᑦ ᓇᐃᓈᖅᓯᒪᔪᑦ

ᐅᑯᐊ ᓇᐃᓈᖅᓯᒪᔪᑦ ᓇᓐᓄᒃᑕᐅᓯᒪᔪᑦ ᐃᓛᒃᑯᑦ ᓇᓄᕐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᐃᓚᒋᔭᐅᓪᓗᓂ ECCC−ᑯᑦ ᑎᒥᖏᓐᓂᑦ ᖃᐅᔨᓴᕋᒃᓴᑦ ᐅᓂᒃᑳᓂᑦ (McGeachy et al. 2023).

ᐱᖓᓱᑦ ᑕᓯᐅᔭᕐᔪᐊᒥᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ, ᐃᓂᖓ 1 ᓂᐅᓪᓴᓐ ᑰᖓ ᓄᓇᕗᑦ/MB ᑭᓪᓕᖓ, ᐃᓚᖓ 2 ᓂᐅᓪᓴᓐ ᑰᖓᓂᑦ WH/SH ᓇᓄᖅᑕᓕᒃ ᑭᓪᓕᖓ ᐊᒻᒪ ᐃᓚᖓ 3 WH/SH ᑭᓪᓕᖓ ᔭᐃᒻᔅ ᐸᐃᒧᑦ (ᐊᔾᔨᓐᖑᐊᖅ 1). ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᐃᓂᐅᔪᒥᑦ 1 2017-2022,ᐃᓂᒋᔭᐅᔪᒥᑦ 2 2017-2022-ᒥᑦ ᐊᒻᒪᓗ ᐃᓂᖓ 3 2021/2022-ᒥᑦ. 2020−ᒥ ᖃᐅᔨᓴᓚᐅᙱᑦᑐᑦ. ᑲᑎᓪᓗᒋᑦ 613 (32%) ᖃᐅᔨᓴᕋᒃᓴᑦ ᑲᑎᖅᓱᖅᑕᐅᓚᐅᖅᐳᑦ ᐃᓂᖓ 1, 724 (38%) ᐃᓂᖓ 2 ᐊᒻᒪ 571 (30%) ᐃᓂᖓ 3.

ᒪᖃᐃᑦᑕᖅᓄᑦ ᖃᐅᔨᓴᕋᒃᓴᑦ (N=377, WH = 187, SH = 190) ᑲᑎᑕᐅᓚᐅᖅᑐᑦ ᓄᓇᕗᒻᒥ ᑯᐸᐃᖕᒥᓪᓗ 2017/2018−ᒥᑦ 2022/2023−ᒧᑦ ᓇᓐᓄᕐᓇᒥᑦ. ᑖᒃᑯᓇᙵᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ, 39 (10%) ᐊᔾᔨᖃᓚᐅᖅᐳᑦ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᓂᒃ 2017-2022 ᐊᑯᓐᓂᖓᓂ (ᐊᔾᔨᙳᐊᖅ 2). ᐃᓂᖓ 1 ᑭᒡᒐᖅᑐᐃᕗᖅ 54%-ᖓᓂ ᐅᑎᖅᑐᒃᓴᓂᑦ, ᐃᓂᐅᔪᖅ 3 ᑭᒡᒐᖅᑐᐃᕗᖅ 28%-ᒥᑦ ᐊᒻᒪᓗ ᐃᓂᖓ 2 ᑭᒡᒐᖅᑐᐃᕗᖅ 18%-ᒥᑦ.

ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐃᓂᐅᔪᒥᑦ 1-ᒥᑦ ᓇᓐᓄᒃᑕᐅᑐᐃᓐᓇᓚᐅᖅᑐᑦ ᐱᖓᖕᓇᕐᒥᑦ ᐱᖓᖕᓇᕐᒥᑦ (WH)-ᒥᑦ ᐊᒻᒪᓗ 81%-ᖑᑉᓗᑎᒃ ᐱᖓᖕᓇᕐᒥᑦ ᐱᖓᖕᓇᕐᒥᑦ ᐱᔭᐅᔪᑦ. ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐃᓂᒥᑦ 2 ᓇᓐᓄᓚᐅᖅᑐᑦ ᑕᒪᐃᓐᓂᒃ WH ᐊᒻᒪ SH. ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᐃᓂᖓ 2−ᒥᑦ 11%−ᖑᓚᐅᖅᐳᑦ ᐅᑎᖅᑐᕚᓃᑦ WH−ᒥᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓪᓗᑎᒃ ᑕᒪᐃᓐᓂ ᐃᓂᒥᑦ 1 ᐊᒻᒪ 2−ᒥᑦ 8%−ᒥᑦ ᓇᓐᓄᒃᑕᐅᓚᐅᖅᑐᑦ. ᑕᐃᒪᐃᒻᒪᑦ, ᑲᑎᓪᓗᒋᑦ, 19%−ᑐᐃᓐᓇᕐᒥᑦ ᐊᖑᔭᐅᔪᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ WH−ᒥ ᐃᓚᓕᐅᔾᔭᐅᓚᐅᖅᐳᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐃᓂᖓ 2−ᒥ.

SH−ᒥ, 85% ᐊᖑᔭᐅᓚᐅᖅᑐᑦ ᓇᓄᕐᓂ ᖃᐅᔨᓴᖅᑕᐅᓵᓚᐅᖅᑐᑦ 3−ᒥ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓪᓗᑎᒃ ᐃᓂᖓᓂ 2 ᑭᒡᒐᖅᑐᐃᓪᓗᑎᒃ 15%−ᒥᒃ.

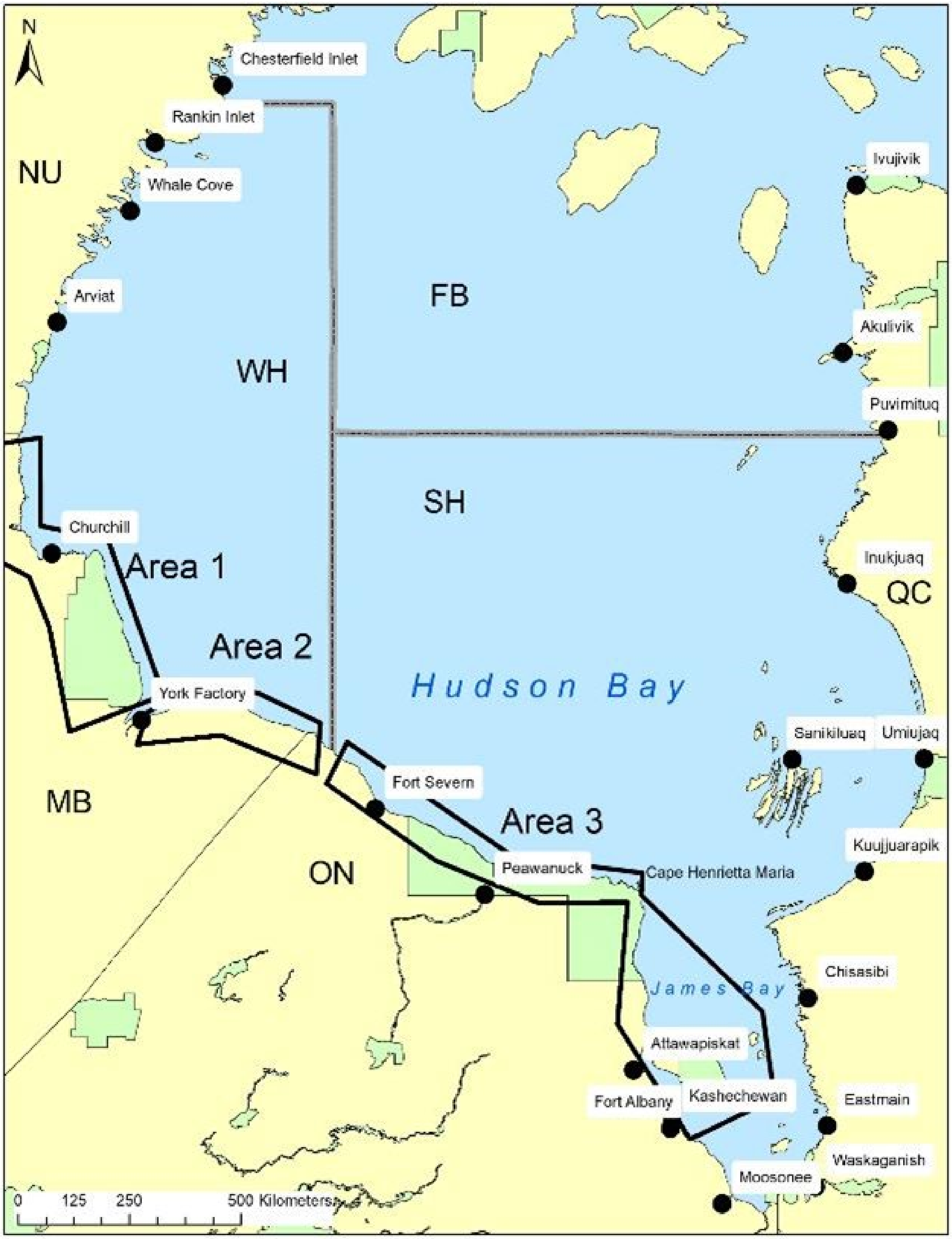
ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᐃᑦ ᓇᓗᓇᐃᖅᐳᖅ ᓇᓄᐃᑦ ᐃᓂᐅᔪᒥᑦ 2-ᒥᑦ ᑲᖏᖅᓱᐊᓘᑉ ᐃᓗᐊᓂ ᓇᓐᓄᒐᔪᙱᓐᓂᖅᓴᐅᔪᑦ. ᐊᖑᓇᓱᒃᑕᐅᒑᖓᑕ, ᐊᖑᓇᓱᒃᑕᐅᓲᑦ ᑕᒪᐃᓐᓂᒃ WH ᐊᒻᒪ SH−ᒥ. ᑕᐃᒪᐃᒻᒪᑦ, ᐃᓱᒪᒋᔭᐅᔪᑦ ᓇᓄᓕᒫᑦ ᐃᓂᒥᑦ 2−ᒥᑦ ᐊᑐᐃᓐᓇᐅᔪᑦ ᐊᖑᓇᓱᖕᓂᖕᒧᑦ ᐱᖓᖕᓇᖓᓂᑦ ᐱᖓᖕᓇᖓᓂᑦ ᐊᑐᖅᑰᔨᙱᑦᑐᑦ. ᐅᔾᔨᖅᓱᑦᑎᐊᕐᓗᑎᒃ ᐱᓕᕆᔭᕆᐊᖃᖅᐳᑦ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑎᖃᓕᕌᖓᑕ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑰᑕᖏᓐᓂᒃ WH−ᒧᑦ ᐱᖃᓯᐅᔾᔨᔪᓂᒃ ᐅᓄᕐᓂᖏᓐᓂᒃ ᒥᒃᓴᐅᑦᑎᓯᒪᔪᓂᒃ ᐃᓂᖓ 2-ᒥ.

ᐅᓂᒃᑳᖅ ECCC−ᑯᓐᓂ (McGeachy et al. 2023) ᑕᑯᒃᓴᐅᑎᑦᑎᓚᐅᖅᐳᖅ ᓇᒧᙵᐅᕙᓪᓕᐊᓂᖏᓐᓂᒃ ᖃᓂᒋᔮᓃᑦᑐᑦ ᓇᓄᐃᑦ ᐱᔾᔪᑕᐅᓪᓗᐊᑕᓚᐅᖅᐳᖅ ᐊᒃᑐᐃᔾᔪᑕᐅᓪᓗᓂ ᐅᓄᕐᓂᖏᓐᓂᒃ ᒥᒃᓴᐅᑦᑎᓯᒪᔪᓂᒃ ᐱᔭᐅᔪᓂᒃ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ 2021−ᒥ WH ᐊᒻᒪ SH.−ᒥ ᓇᒧᙵᖅᐸᓪᓕᐊᓂᖏᑦ

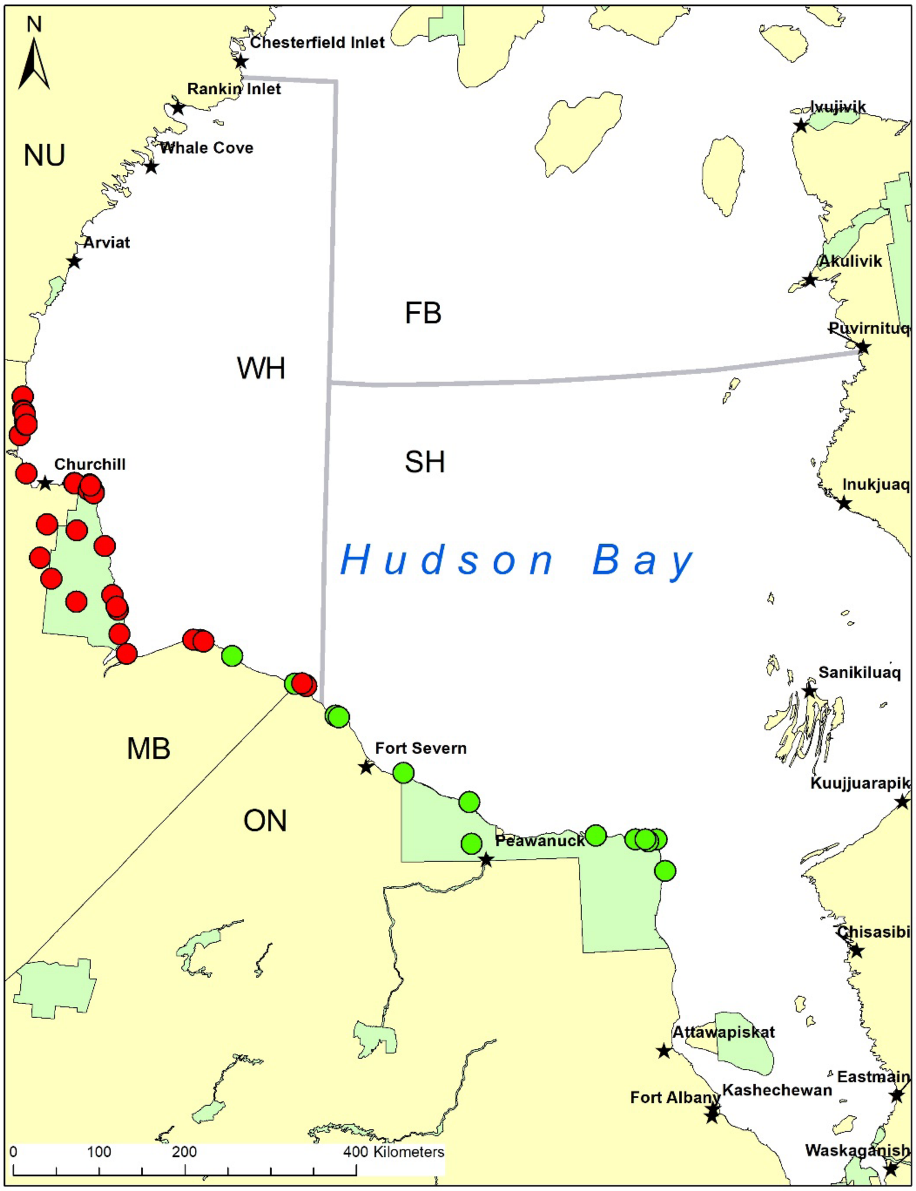
ᖃᓂᒋᔮᓃᑦᑐᑦ ᓇᓄᐃᑦ ᐱᔾᔪᑕᐅᓪᓗᐊᑕᓚᐅᖅᐳᖅ ᑕᐅᖅᓰᖃᑎᒌᖕᓂᕐᒧᑦ ᐃᓂᒥᒃ 2 ᐊᒻᒪ ᐃᓂᖓ 3 ᐊᒃᑐᐊᓂᖃᖅᕈᓂ ᓱᑯᖃᕐᓂᖓ. ᐃᓱᒪᒋᔭᐅᑎᓪᓗᒍ ᐊᑕᐅᑦᑎᒃᑯᑦ ᑖᒃᑯᓄᖓ ᓇᐃᓈᖅᓯᒪᔪᓄᑦ, ᓇᓄᐃᑦ ᓇᒧᙵᖅᐸᓪᓕᐊᓂᖏᑦ WH ᐊᒻᒪ SH ᐊᑯᓐᓂᖓᓂ ᐃᓄᐃᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᙱᓐᓂᖅᓴᐅᔪᑦ ᐊᖑᓇᓱᒋᐊᒃᓴᖅ WH ᐊᒻᒪ SH−ᒥ.

ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ

ᒪᒌᓯ, ᑎ., ᓚᓐ, N.J., Northrup, J.M., Trim, V., Davis, C., ᐊᒻᒪ Derocher, A.E. (2023). ᓇᓅᖃᑎᒌᑦ ᓇᓅᖃᑎᒌᑦ (*Ursus maritimus*) ᑲᖏᖅᓱᐊᓘᑉ ᐃᓗᐊᓂ ᓱᑯᖃᕐᓂᖓ ᐱᔾᔪᑎᒋᓪᓗᒍ, 2017-2022 ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᖅ. ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ, 32 pp.



ᐊᔾᔨᖑᐊᖅ 1. ᖃᐅᔨᓴᕐᕕᐅᔪᑦ ᑕᑯᒃᓴᐅᑎᑦᑎᕗᑦ ᐱᖓᓱᓂᒃ ᓄᓇᓂᒃ ᐊᕕᒃᑐᖅᓯᒪᔪᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ (Area 1, ᐃᓂᖓ 2 ᐊᒻᒪ ᐃᓂᖓ 3). ᐃᓂᖓ 1 ᐊᒻᒪ ᐃᓚᖓ 2 ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ 2017-2021 ᐊᒻᒪ ᐃᓚᖓ 3 ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᖅ 2021-2023. 2020−ᒥ ᖃᐅᔨᓴᓚᐅᙱᑦᑐᑦ.



ᐊᔾᔨᙳᐊᖅ 2. ᓇᒦᓐᓂᖏᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ 2017-2022−ᒥ ᑭᖑᓂᐊᒍᑦ ᓇᓐᓄᒃᑕᐅᓚᐅᖅᑐᑦ WH−ᒥ (ᐊᐅᐸᖅᑐᑦ ᐊᖕᒪᓗᖅᓯᓯᒪᔪᑦ) ᐅᕝᕙᓘᓐᓃᑦ SH (ᐆᔭᐅᔭᖅ ᐊᖕᒪᓗᖅᓯᓯᒪᔪᑦ) 2017/18 ᐊᒻᒪ 20122/23 ᓇᓐᓄᑲᓯᖕᓇᒥ.

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ᓇᒧᖔᓯᒪᓂᖕᒥᑦ ᓇᓄᐃᑦ (*Ursus maritimus*) ᑕᓯᐅᔭᕐᔪᐊᒥ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᖅ ᓯᑯᐊ ᐊᓯᖑᖅᓯᒪᓕᕐᒪᑦ, 2017-ᒥᑦ 2022-ᒧᑦ ᑭᖑᓪᓕᖅᐹᖅᓯᐅᑎ ᐅᓂᒃᑳᖅ

ᑎ. ᒥᒃᔨᑦᓯ[1,4], ᐃᓐ.ᔨ. ᓚᓐ [1], ᔨ. ᐃᒻ. ᓄᐊᑐᓚᑉ [2], ᕕ. ᑐᓕᒻ [3], ᓯ. ᑕᐃᕕᔅ[4], ᐊ. ᐃ ᑐᕈᑦᓱ [4]

[1] ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚᐅᑉ ᐅᕐᑰᓯᕙᓪᓕᐊᓂᖓᑕ ᐊᓯᕈᕐᓂᖓ ᑲᓇᑕ, CW-ᐆᒪᔪᕐᓂᒃ ᖃᐅᔨᓴᖅᑎᒻᒪᕇᑦ ᐃᒡᓗ,
ᐃᓕᓐᓂᐊᕐᕕᒃᔪᐊᖅ ᐊᐃᐳᑕ, ᐃᒻᒪᑕᓐ, ᐊᐃᐳᑕ T6G 2E9
[2] ᐋᓐᑎᐅᕆᐅ ᒥᓂᔅᑕ ᓄᓇᒥᐅᑕᓕᕆᔨᒃᑯᓄᑦ ᐊᒻᒪᓗ ᓇᐹᖅᑐᖃᖅᑐᓄᑦ, ᐊᐅᖕᑎᒍᑦ ᖃᐅᔨᓴᕐᕕᒃ ᐃᒡᓗ B217L ᑐᓕᑦ
ᐃᓕᓐᓂᐊᕐᕕᒃᔪᐊᖅ,2140 East Bank Drive, ᐱᑐᐳᓗ, ᐋᓐᑎᐅᕆᐅ K9L 1Z8
[3] ᒫᓂᑑᐸ ᐊᐅᓚᑦᓯᕕᖓ ᓄᓇᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᐅᑭᐅᖅᑕᖅᑐᒥ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᖅ, ᐃᒪᕐᒥᒃ ᐸᕿᔮᑦ ᐊᒻᒪᓗ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ
ᐊᐅᓚᑦᓯᕕᖓ, ᓂᕐᔪᑎᑦ ᐊᒻᒪᓗ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐊᐅᓚᑦᓯᕕᖓ, ᑎᑎᖅᑲᒃᑯᕕᒃ 28, 59 Elizabeth Drive, ᑕᒻᓴᓐ, ᒫᓂᑑᐸ R8N 1X4
[4] ᐊᐅᓚᑦᓯᕕᖓ ᐆᒪᔪᓂᒃ ᖃᐅᔨᓴᖅᑎᒻᒪᕇᑦ, ᐃᓕᓐᓂᐊᕐᕕᒃᔪᐊᖅ ᐊᐃᐳᑕ, ᐃᒻᒪᑕᓐ, ᐊᐃᐳᑕ T6G 2E9

ᐊᑏ ᐃᓕᓴᓇᓗᒍ ᐅᖃᓕᒫᒐᖅ: ᒥᒃᑎᑦᓯ, ᑎ., ᓚᓐ, ᐃᓐ.ᔫ., ᑐᓕᒻ, ᕃ., ᑕᐃᕃᔅ. ᓯ., ᐊᒻᒪᓗ ᑐᓛᑦᓱ, ᐊ, ᐃ. (2023). ᓇᒧᖕᓛᕙᖕᒪᖕᓛᑕ ᐊᓯᕈᖅᑐᑦ ᓇᓄᐃᑦ (*Ursus maritimus*) ᑕᓯᐅᔭᕐᔪᐊᒥ ᒪᓕᒃᑐᑦ ᑕᓇᐅᖅ ᖃᓄᓇᑑᓂᖕᒋᓐᓂᒃ, 2017-2022 ᑭᖕᒍᓪᓕᖅᐹᖅᓯᐅᑎ ᐅᓂᒃᑳᖅ, ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪᓗ ᓯᓚ ᐅᖅᑰᓯᕚᓪᓕᐊᓂᖕᒐᓄᑦ ᐊᓯᕈᖅᑐᖅ ᑲᓇᑕ, 32 ᒪᒃᐱᒐᖅ.

ᐊᐅᓚᑦᓯᔨᓂᑦ ᓇᐃᓈᖅᓯᒪᔪᖅ

ᖃᐅᔨᓴᖅᐳᒍᑦ ᓇᒧᖔᓯᒪᖕᒪᖔᑕ ᐊᓯᕈᕐᒪᖔᑦ ᐊᒻᒪᓗ ᐃᓚᒃᑲᓐᓂᖏᑦ ᓇᓄᐃᑦ ᑖᒃᑯᓇᖔᖅᑐᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᖑᓯᒪᔪᓂᑦ ᑕᕆᐅᖓᓂ ᖃᓂᒋᔭᖓᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᑭᒡᓕᐊᓂᑦ ᐊᕙᑖᓂᑦ 2017 ᐊᒻᒪᓗ 2022. ᐊᑐᖅᓯᓚᐅᖅᑐᒍᑦ ᖃᓄᐃᓕᐅᕈᑎᑦ ᖃᐅᔨᓴᖅᓚᐅᖅᑕᑦᑕ ᓇᓄᐃᑦ ᑭᖑᓕᖏᑦ ᓇᒧᖔᓯᒪᕙᖏᑦ ᐃᓂᖏᑦ ᐱᓪᓗᒋᑦ ᐊᒥᐊᒃᑯᑦ ᓯᑯᓂ ᐃᓗᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᓇᒧᖔᓯᒪᓂᖏᑦ ᐊᒻᒪᓗ ᖃᓄᖅ ᐊᔾᔨᒌᙱᑦᑑᑕᐅᖕᒪᖔᑕ ᖃᐅᔨᓴᕐᓗᒋᑦ ᐊᕐᕌᒍᑕᒫᑦ ᑕᒫᓂᕐᓇᔪᑦ ᑕᐃᒫᒃ ᐊᔾᔨᖏᓐᓂᒃ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᒋᓗᒋᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ, ᐃᓐᓇᕈᓯᔪᑦ ᐊᕐᓇᐃᑦ ᐊᒻᒪᓗ ᐃᓐᓇᕈᓯᔪᑦ ᐊᖑᑎᑦ.

ᑕᐃᑲᓂ ᐱᒋᐊᓕᖅᑎᓪᓗᒍ ᐃᓕᑦᑎᓇᓱᒃᑐᑦ, ᑕᒪᕐᒥᐸᓗᒃ ᓇᓄᐃᑦ ᐃᓚᖏᓐᓂ ᑭᒡᓕᐊᓃᓚᐅᖅᑐᑦ ᐱᓕᕆᔭᐅᓚᐅᙱᑦᑐᑦ ᐊᒻᒪᓗ ᓂᕕᖓᑖᓕᖅᑕᐅᓚᐅᙱᑦᑐᑦ ᖃᐅᔨᓴᖅᑎᓂᑦ (57-71% ᐊᕙᑖᓂᑦ 2017 ᐊᒻᒪᓗ 2019), ᓂᕕᖓᑖᓕᖅᑕᐅᖅᓯᒪᓪᓗᑎᒃ ᓇᓄᐃᑦ ᑲᑎᖓᔪᑦ ᑖᒃᑯᓄᖓ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (72%) ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (28%) ᓂᕐᔪᑎᑦ. 2021-ᒥ, ᑕᐃᑲᓂ ᑕᒪᒃᑭᑦᑎᐊᕋᓱᒃᑎᓪᓗᒋᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᐃᓪᓗᑎᒃ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᖃᖓᑕᓲᒃᑯᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ, ᓯᑯᓚᒫᖕᓂ ᐃᓗᐊᓂ ᑭᓪᓕᖏᑦᑕ ᑖᒃᑯᐊ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ, ᑕᐃᑲᖓᑦ ᓯᒡᔭᖓᑕ ᐱᒍᐊᓄᑦ, ᐋᓐᑎᐅᕆᐅ. ᖃᐅᔨᓴᕈᑎᒃᓴᖅᑖᓚᐅᖅᑐᒍᑦ 183 ᐊᑐᓂ ᓇᓄᕐᓂᒃ ᑕᐃᑲᖓᖅ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑖᒃᑯᓇᖓᑦ 25% ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᓐᓂᒃ, 10% ᑖᒃᑯᓄᖓ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᐊᒥᐊᒃᑯᓪᓗᑎᒃ 65% ᓯᕗᓪᓕᕐᒥ ᐱᓕᕆᐊᖑᔪᓂ. ᑕᐃᑲᓂ 2022 ᓯᑯᓚᒫᓕᓚᐅᖅᑐᖅ ᑕᒪᕐᒥᐸᓗᒃ ᒫᓂᑑᐸ ᐊᒻᒪᓗ ᐋᓐᑎᐅᕆᐅ ᓯᓪᕗᖏᑦ ᐊᒻᒪᓗ ᓯᑯᖅᑲᕐᓂᖅᓴᐅᓪᓗᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐊᔾᔨᒋᓇᓂᐅᑉ 2021-ᒥᑦ. ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ 73 ᓇᓄᕐᓂᑦ ᑕᐃᑲᖓᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᐅᑯᐊᓗ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᓚᐅᖅᑐᑦ ᑕᐃᑲᓂ 2022; 64%-ᖏ ᑖᒃᑯᓇᖓᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᖅᑐᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ 36% ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ. ᖃᓄᖅ ᐊᔾᔨᒌᙱᑎᒋᖕᒪᖔᑕ, ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ 120 ᓇᓄᕐᓂᒃ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ 2021-ᒥ ᐅᑯᐊᓗ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᓚᐅᖅᑐᑦ 2022-ᒥ; 93% ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐊᒻᒪᓗ 7% ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᐅᓇ ᐱᓯᒪᓂᖅᑲᖅᑐᖅ ᐃᒻᒪᖄ ᐅᓄᖅᓯᒋᐊᖅᑐᑦ ᓇᓄᐃᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᑕᐃᑯᖓ 29% ᐊᒻᒪᓗ ᑕᐃᒫᓪᓗᐊᖅᑕᐅᖅ ᐅᓄᕈᓐᓃᕆᕈᓪᓗᑎᒃ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᑲᖓᑦ 2021 ᑎᑭᓪᓗᒍ 2022 ᒪᓕᒃᑐᑦ ᓄᒃᑕᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᖅᓯᕗᑦ ᓈᓴᖅᓯᒪᔭᑦᑎᒍᑦ.

ᑕᕆᐅᖅ ᓯᑯᐊ ᓄᑦᑕᖅᑎᑦᑎᓲᖅ ᓇᓄᕐᓂᒃ ᑕᓪᓕᒪᐃᒃᑲᓐᓂᓪᓗᐊᖅᖢᒍ ᓅᑦᑕᕋᔭᖅᑐᑦ ᐱᓇᖕᓇᖅᐸᓯᖓᓄᑦ ᐊᕐᕌᒍᓂ ᑕᐃᑲᓂ ᓯᑯᒐᓛᕐᑲᓕᕌᖓᑦ ᐱᓇᖕᓇᖓᓂ ᐃᑉᐱᐊᕐᔪᐊᑕ. ᐃᓐᓇᖅ ᐊᖑᑎᑦ ᓇᒧᖓᐅᔾᔪᓯᖕᓂᑦ ᐊᔾᔨᒋᔪᓐᓃᖅᑕᖕᓂᑦ ᐊᕙᑖᓂ ᐊᕐᕉᒍᑦ ($F_{2, 222}$ =4.175, P= 0.02) ᖃᖓᐅᓕᕉᖓᑦ ᓅᖃᑦᑕᕐᓂᓴᐅᓕᓲᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᖏᓂᖅᓴᐅᓕᕐᒧᑦ ᐊᕐᕉᒍ ᐃᓗᐊᓂ ᐊᓯᕈᐊᑦᑕᖅᑐᑦ ᓯᑯᐃᑦ. ᐃᓚᖏᑦ ᐊᕐᓇᐃᑦ ᓇᒧᖔᐅᑲᐅᓂᖅᓴᐅᔪᑦ ᑖᒃᑯᓇᖓᓪᓕ ᐃᓐᓇᕐᓂᑦ ᐊᕐᓇᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐃᓐᓇᕐᓂᑦ ᐊᖑᑎᓂᑦ ($F_{3, 185}$ = 3.57, P= 0.02) ᑭᓯᐊᓂ ᖃᓅᑎᒋᖕᒪᖕᒫᑕ ᖃᐅᔨᓇᓱᒃᓯᒪᓇᑎᒃ ᐃᒻᒥᖕᓄᑦ ᑭᓯᐊᓂ ᐊᓯᖏᓐᓅᓐᖏᑦᑐᖅ ᐃᓚᖏᓐᓄᑦ ᐃᓐᓇᕐᓄᑦ ᐊᖑᑎᓄᑦ. ᐊᑐᓂ ᐃᒻᒥᒍᑦ ᐱᕚᓪᓕᕈᑎᖃᖅᑕᖅᑐᖅ ᖃᓄᑐᐃᓐᓇᖅ ᓇᒧᖔᓯᒪᓂᖏᑦ ᐊᖏᔫᖅ 87%, 66%, 77% ᐊᒻᒪᓗ 77% ᐃᓐᓇᕐᓄᑦ ᐊᖑᑎᓄᑦ, ᐃᓐᓇᕐᓄᑦ ᐊᕐᓇᓄᑦ, ᐃᓚᖏᓐᓄᑦ ᐃᓐᓇᕐᓄᑦ ᐊᖑᑎᓄᑦ, ᐃᓚᖏᓐᓄᑦ ᐃᓐᓇᕐᓄᑦ ᐊᕐᓇᓄᑦ, ᐊᑐᓂ.

ᑎᑎᕋᖅᓯᓚᐅᖅᑐᒍᑦ ᐊᓄᕐᓂᖅᓴᐅᔪᑦ ᑭᐳᒃᑐᑦ ᐊᕙᑖᓂᑦ ᓴᓂᓕᕆᔭᒥᖕᓂᑦ ᐃᓚᒌᑦ ᓇᓄᐃᑦ ᑕᐃᒪᖓᓂᑦ ᐅᖃᐅᓯᐅᕙᓕᕐᒪᑕ ᐅᓄᕐᓂᖅᓴᖏᑦ ᓇᒧᖐᖃᑦᑕᖅᑐᑦ ᑕᒪᕐᒥᒃ ᐅᑭᐅᖏᑦᑎᒍᑦ ᐊᒻᒪᓗ ᐊᕐᓇᐃᑦ ᐊᖑᑎᑦ ᐊᒻᒪᓗ ᐊᑕᔪᑦ ᖃᓄᕆᑑᓂᖓᓄᑦ ᑕᕆᐅᖅ ᓯᑯᐊ. ᓇᓂᔭᑦᑎᒍᑦ ᐃᓱᒪᓕᖅᓯᔫᖅ ᕿᓚᒥ ᓇᒧᖐᖃᑦᑕᖅᑐᑦ ᓴᖑᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ, ᐱᓗᐊᖅᑐᒥᒃ ᖃᓂᒋᔭᖏᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ/ᐊᑭᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦᑕ ᑭᒡᓕᖏᑦ ᐅᓄᖅᓯᔪᑦ ᖃᐅᔨᒪᓇᖑᑦᑐᒧᑦ ᖃᓄᖅ ᐅᓄᖅᑎᒋᖕᒪᖕᒫᑕ ᐊᒻᒪᓗ ᐊᓯᕈᖅᑕᖃᑦᑕᕐᓂᖏᑦ ᑖᒃᑯᐊ ᖃᑦᓯᐅᓇᓱᒋᔭᐅᓂᖏᑦᑕ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᓂ ᐊᑐᓂ ᓇᓄᐃᑦ ᐊᕗᑎᓯᒪᔪᑦ. ᑕᐃᒪᓗ ᖃᐅᔨᒪᓐᖏᓂᕗᑦ ᐱᒻᒪᕆᐅᔪᖅ ᐊᒃᑐᐃᔪᓐᓇᕐᒪᑕ ᐊᐅᓚᑕᐅᓂᖏᓐᓂᒃ ᓇᓄᐃᑦ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ.

ᐱᒋᐊᕐᓂᖓ

ᓇᓄᐃᑦ ᓇᒧᖕᒫᑎᓲᑦ ᑕᒪᐅᓐᓇ ᓯᑯᒃᑯᑦ ᐃᒫᒍᑦ ᓯᓚᕐᔪᐊ ᑲᔾᔨᐊᓃᑦᑐᑦ ᐅᑭᐅᖅᑕᖅᑐᒥ 19 ᖃᑦᓲᓇᐃᑦ ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᐊᑐᖅᓯᓪᓗᑎᒃ ᑭᓯᐊᓂ ᑕᕆᐅᖅ ᓯᑯᐊᓂ ᐃᓂᖃᖅᓗᑎᒃ ᑕᐃᑲᖕᖓᓪᓗ ᓇᑦᑎᕋᓱᒃᐸᒃᓗᑎᒃ, ᑖᒃᑯᐊ ᓂᕿᓪᓗᐊᑕᕆᖕᒪᕆᑦ (ᓯᑑᓕᖕ ᐊᒻᒪᓗ ᐅᕆᑦᓯᓚᓐᑦ 1995, ᓚᐃᑐᓕ ᐊᓯᖏᓪᓗ. 2008, ᑏᒪᓐ ᐊᓯᖏᓪᓗ. 2008, ᑲᓕᓯᐊ ᐊᓯᖏᓪᓗ. 2015). ᑕᐃᒪᓐᓇᐅᒐᓗᐊᖅᑎᓪᓗᒍ ᓇᓄᐃᑦ ᐅᐸᒃᓯᓯᒪᐃᓐᓇᓲᑦ ᑕᒫᓂᑦᑕᐃᓐᓇᐸᓗᒃ ᑎᑭᐅᑎᔪᓐᓇᖅᑕᒥᓄᑦ ᐃᓕᖁᓯᑐᖃᖏᑦ, ᐅᑭᐅᖅᑕᖅᑐᖅ ᐅᖁᓯᕙᓪᓕᐊᖕᒪᑦ ᓱᒃᑲᓕᔪᒥᒃ ᑕᐃᒪᓗ ᓯᓚᕐᔪᐊᓕᒫᑎᒍᑦ ᐊᒻᒪᓗ ᓄᖑᓕᖅᐳᖅ ᐅᑭᐅᖅᑕᖅᑐᖅ ᑕᕆᐅᖕᒪᑕ ᓯᑯᐊ ᓱᒃᑲᓕᔾᒪᕆᐋᓗᖕᒥᒃ (ᓯᑐᓗᕝ ᐊᓯᖏᓪᓗ. 2012, ᐅᓚᓐᑕᓐ ᐊᓯᖏᓪᓗ. 2022). ᐱᑕᖃᕈᓐᓃᖅᐸᑦ ᑕᕆᐅᖅ ᓯᑯᐊ, ᑕᐃᒪᐃᑎᑕᐅᓗᓂ ᐃᓄᖕᓂᑦ ᐊᓯᕈᖅᑎᑦᑎᔪᑦ ᓯᓚᐅᑉ ᖃᓄᐃᔾᔪᑎᖕᒐᓂᒃ, ᐅᑯᐊ ᐅᖃᐅᓯᖃᖅᓯᒪᔪᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ ᓴᐳᓐᓂᐊᕆᐊᖃᕐᓂᖏᓐᓄᑦ ᐃᓱᒫᓘᑎᑦ ᐅᑭᐅᖅᑕᖅᑐᒥ ᐳᐃᔨᓄᑦ, ᑖᒃᑯᓄᖕᒐᓗ ᓇᓄᕐᓄᑦ. (ᓯᑑᓕᖕ ᐊᒻᒪᓗ ᑐᓚᐅᓯ 1993, ᑐᓚᐅᓯ ᐊᓯᖏᓪᓗ. 2004, ᓚᐃᑐᓕ ᐊᓯᖏᓪᓗ. 2015).

ᐅᑯᐊ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ ᐱᑕᖃᓲᑦ ᓂᒋᐊᓂ ᐅᖓᓯᓗᐊᕋᑎᒃ ᐆᒪᔪᓄᑦ. ᐅᓄᖅᑐᑦ ᐅᐱᕐᖕᒑᒃᑯᑦ ᐅᖁᓯᑎᓪᓗᒍ ᓄᓇᖕᒐᓐᓂ ᕿᓚᒥ ᑕᕆᐅᖅ ᓯᑯᐊ ᐊᐅᖃᑦᑕᖅᑐᖅ ᓯᑯᐃᖃᑦᑕᖅᑐᖅ ᐊᒻᒪᓗ ᑭᖑᕚᖅᓯᒪᓪᓗᓂ ᖁᐊᖅᑕᖅᑐᖅ (ᑳᓂᐅ ᐊᒻᒪᓗ ᑲᐅ 2005, ᓯᑑᓕᖕ ᐊᒻᒪᓗ ᐹᑭᓐᓯᓐ 2006, HᐅᑦᔅHᐃᒻ ᐊᓯᖏᓪᓗ. 2010). ᐅᑯᐊ ᐊᓯᕈᖅᑐᑦ ᐊᑯᓂᐅᑎᒋᓂᖕᒐ ᓇᓄᐃᑦ ᓇᑦᑎᕋᓱᒍᓐᓇᕐᓂᖕᒐᑕ ᓯᑯᒥ, ᐱᓗᐊᖅᑐᒥᒃ ᐊᑦᑕᕐᓇᖅᓯᑎᓪᓗᒍ ᐅᐱᕐᖕᒑᒃᑯᑦ, ᐊᒻᒪᓗ ᐊᔭᒃᓯᕗᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᕐᓂᒃ ᐊᑯᓂᐅᓂᖅᓴᖅ ᓄᓇᒦᖃᑦᑕᓕᖅᐳᑦ, ᑕᐃᑲᓂᓗ ᓈᒻᒪᕆᖕᖏᑕᒥᖕᓃᑉᐳᑦ ᑎᒥᖕᒐᓄᑦ (ᓯᐅᓕ ᐊᓯᖏᓪᓗ. 2013, ᓚᓐ ᐊᓯᖏᓪᓗ. 2016, ᔮᓯᓐ ᐊᓯᖏᓪᓗ. 2020). ᐊᑯᓂᐅᓂᖅᓴᖅ ᓯᑯᖃᖕᖏᒻᒪᑦ ᐊᑕᔪᖅ ᑎᒥᖏᑦ ᓴᓪᓗᐊᓘᓕᕈᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ ᕿᑐᖕᖓᒃᓴᖃᕐᓂᖅᓴᐅᓗᑎᒃ ᐊᑭᓐᓇᖅᐸᓯᐊᓂᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ (ᑐᓚᐅᓯ ᐊᒻᒪᓗ ᓯᑑᓕᖕ 1995b, ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 1999, ᓯᑭᐅᓪᓗ ᐊᓯᖏᓪᓗ. 2016). ᒫᓐᓇᓵᖅ, ᐅᓄᕈᓐᓃᖅᑐᑦ ᐆᒪᔾᔪᑎᒃᓴᖃᕈᓐᓃᕋᒥᒃ ᐊᒻᒪᓗ ᑕᕆᐅᖅ ᓯᑯᐊ ᑕᐃᒪᐃᑎᑦᑎᔪᖅ (ᐅᓕᔅHᐅ. 2007, ᓚᓐ ᐊᓯᖏᓪᓗ. 2016). ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᑯᓚᐃᑦᑐᒥᒃ ᒥᐊᓂᕆᔭᐅᓐᖏᓂᖅᓴᐅᓕᖅᒪᑕ ᓇᓄᐃᑦ ᑕᐃᒪᖕᒪᒥᑦ ᐊᓂᒍᖅᑐᑦ 40 ᐊᕐᕌᒍᓂ, ᑭᓯᐊᓂ ᑕᑯᓯᒪᔪᑦ ᑕᐃᒫᑦᑎᐊᖅ ᐊᑐᖅᐸᒃᑲᖏᑦ, ᐊᑯᓂᐅᓂᓴᐅᔭᕌᖕᒪᑕ ᓯᑯᖃᖕᖏᓂᖕ ᐊᕐᕌᒍᒥ (ᓯᑑᓐ ᐊᒻᒪᓗ ᓚᐃᑐᓕ, 2016), ᐊᒻᒪᓗ ᑎᒥᖏᑦ

ᓱᒃᑯᑉᐸᓪᓕᐊᔪᑦ ᐊᒻᒪᓗ ᐆᒪᓇᓯᒐᑐᐃᓐᓇᓲᑦ ᐊᕙᓘᓂ 1980 ᐊᒻᒪᓗ ᐱᒋᐊᕐᓂᖓᓂ 2000 (ᐊᑉᐹᑦ ᐊᓯᖏᓪᓗ. 2007, 2016).

ᓇᒧᖕᒫᓯᒪᓂᖏᑦ, ᐅᓄᕐᓂᖏᑦ, ᐊᒻᒪᓗ ᕿᒪᒍᑎᕙᓪᓕᐊᔪᑦ ᑭᒡᓕᖏᓐᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ (ᓴᖅᑭᔮᖅᑐᖅ 1) ᐱᔾᔪᑎᖃᖅᑐᑦ ᓇᓐᓄᒃᑕᐅᔪᓄᑦ ᐊᒻᒪᓗ ᐱᔭᐅᒃᑲᓐᓂᖅᑐᑦ ᓂᕈᖕᒐᑖᓕᑦ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 1977, ᑐᓚᐅᓯ ᐊᒻᒪᓗ ᓯᑑᓕᖕ 1990, ᑐᓚᐅᓯ ᐊᒻᒪᓗ ᓯᑑᓕᖕ 1995a, ᑕᐃᓗ ᐊᒻᒪᓗ ᓖ 1995). ᑕᐃᒪᓐᓇᐅᓚᓗᐊᖅᑎᓪᓗᒍ ᓴᓂᑦᑎᐊᖓ ᓄᓇᖓᓂ ᓯᑯᓯᕚᕐᒥᐅᑦ ᐃᒪᖓ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ ᑲᑎᖕᖓᔭᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᐊᑐᓂ ᑕᕆᐅᖅ ᓯᑯᐊᓂ ᓄᓕᐊᓲᑦ (ᐋᖕᑯᓂ ᐊᓯᖏᓪᓗ. 2016), ᑕᐃᑲᓂ ᓯᑯᖃᖕᖏᓂᓕᒫᖕᖓᓂ, ᐅᓄᓛᑦ ᓇᓄᐃᑦ ᓄᓕᐊᓲᑦ ᐊᐅᔭᕐᕕᒋᓂᐊᖅᑕᒥᖕᓂ ᐊᒻᒪᓗ ᐅᐸᒍᑎᑲᑕᖕᖏᑦᑐᐊᖅᑐᑦ ᓴᓂᒥᖕᓂ ᓇᓄᕐᓄᑦ ᑕᐃᒪᓐᓇᐅᓇᓯᒋᔭᐅᔪᑦ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 1977, ᑐᓚᐅᓯ ᐊᒻᒪᓗ ᓯᑑᓕᖕ 1990, ᐲᑳᒃ ᐊᓯᖏᓪᓗ. 2010, ᐊᑉᐹᑦ ᐊᒻᒪᓗ ᒥᑦᑎᐅ 2012, ᓚᓐ ᐊᓯᖏᓪᓗ. 2016). ᓄᓕᐊᖅᑦᑕᖅᑐᑦ ᐊᐅᔭᕐᕕᒋᓂᐊᖅᑕᒥᖕᓂ ᓄᓇᒥ ᐊᒥᓲᔪᑦ, ᐃᓗᐊᓂ ᐊᕐᕌᒍ ᐊᔾᔨᒌᓐᖏᑑᑎᓪᓗᒍ ᓯᑯᖕᖓ ᑕᑯᑎᑦᑎᓯᒪᔪᖅ ᓇᓄᐃᑦ ᓇᒧᖕᒫᓂᖅᓴᐅᓲᑦ ᓇᕈᔭᑦᑐᑦ ᓯᑯᖃᖕᖏᑎᓪᓗᒍ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 1999, ᓯᐅᓚ ᐊᓯᖏᓪᓗ. 2013). ᐳᕙᕕᑦ ᐊᒻᒪᓗ ᑯᓚᓈᓯᑭ (1982) ᐊᑐᖅᑕᐅᔪᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᑦ ᓯᒡᔭᖏᓐᓂ ᐊᒻᒪᓗ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐊᒥᓱᓂᖅᓴᐅᔪᑦ ᓇᓄᐃᑦ ᐋᓐᑎᐅᕆᐅᒥ (ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ) ᑕᐃᑲᓂ ᐅᓄᓐᖏᓚᐅᖅᑐᑦ ᑕᑯᔭᐅᔪᑦ ᒫᓂᑑᐸᒥ (ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ) ᐊᒻᒪᓗᑦᑕᐅᖅ ᑕᐃᓐᓇ, ᐃᓯᒪᓕᖅᓯᒪᔪᖅ ᐅᓄᕐᓂᖅᓴᐃᑦ ᑕᐅᕝᕙᖕᒫᖃᑦᑕᖅᑐᑦ ᐅᖃᖅᓯᒪᓚᐅᕐᓗᐊᖅᑎᓪᓗᒍ ᑭᖑᓂᑦᑎᓐᓂ ᑕᐃᒪᓐᓇᐅᓇᓯᒋᔭᐅᓪᓗᓂ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 1977). ᑭᖑᓂᐊᒍᑦ ᖃᐅᔨᓴᖅᑐᑦ ᑕᑯᑎᑦᑎᔪᑦ ᑖᒃᑯᐊ ᑕᓇᐅᖅ ᓯᑯᖏᑦ ᑕᐃᒪᐃᑎᑦᑎᔪᑦ ᓇᓄᐃᑦ ᓇᒧᖕᒫᓯᒪᓪᓗᑎᒃ ᐃᓗᐊᓂ ᐃᓚᒌᒃᑕᖅᑐᑦ ᑭᓯᐊᓂ ᐊᕙᓘᓃᑦᑐᐃᓐᓇᖏᑦᑐᖅ ᐃᒻᒥᖕᓂᑦ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 2004). ᐊᑐᖅᓯᓚᐅᖅᑐᒍᑦ ᓈᓚᐅᑎᑦ ᓂᐱᔾᔨᐅᕙᓪᓕᐊᔪᑦ, ᑕᒪᐃᓐᓂᑦ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. (1999) ᐊᒻᒪᓗ ᓯᐅᓚ ᐊᓯᖏᓪᓗ. (2013) ᓇᓂᓯᓚᐅᖅᑐᑦ ᑖᒃᑯᐊ ᐊᒥᐊᒃᑯᑦ ᓯᑯᒐᓛᑦ ᓱᓕ ᐱᓇᖕᓇᖕᒐᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᓴᖑᔪᑦ ᓇᖕᒥᓂᖅ ᓯᒡᔭᒥᖕᓄᑦ ᑎᑭᖃᑦᑕᖅᑐᑦ ᓇᒧᖕᖓᓪᓗᑎᒃ ᑕᐅᓄᖓ ᓂᒋᐊᓄᑦ ᑲᓇᖕᓇᖓᓄᑦ. ᑕᐃᒪᓐᓇᐅᓚᓗᐊᕐᒪᑦ, ᐊᔾᔨᒌᓐᖏᑦᑐᑦ ᓄᓇᒥ ᓇᒧᖕᒫᓯᒪᓂᖏᑦ ᓇᓄᐃᑦ ᓄᓇᖓᓐᓂ ᐊᒻᒪᓗ ᖃᓄᖅ ᐅᓇ ᐊᓯᑦᔨᑎᑎᓂᐊᖅᐸ ᓄᖑᐸᓪᓕᐊᖕᖏᓐᓇᕐᓗᑰᒃ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓂᕿᖏᑦ ᓂᕿᒃᓴᖃᖕᖏᓪᒪᑕ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᐃᓕᑕᐅᓇᓯᓚᐅᖅᑐᓂᑦ

ᓴᓇᓯᒪᔪᑦ ᓈᓴᖅᑕᐅᔾᔪᑎᖏᓐᓂᑦ ᒫᓐᓇ ᓇᓄᐃᑦ ᑭᒡᓕᖏᓐᓃᑦᑐᑦ. ᐅᓇ ᐱᑐᐊᖅᑐᒥᒃ ᐱᒻᒪᕆᐅᔪᖅ ᓄᓇᐃᑦ ᐊᐅᓚᑕᐅᖕᒪᑕ ᐊᑐᖅᓯᓪᓗᑎᒃ ᖃᖓᑕᓲᓂᒃ ᓈᓴᐃᓪᓗᑎᒃ ᑕᐃᑲᓂ ᐅᓄᕐᓂᖏᑦ ᓇᓚᐅᑖᖅᓯᒪᓪᓗᑎᒃ ᐊᑕᐅᓯᕐᒥ ᐊᕐᕌᒍᒥ.

ᑕᐃᑲᓂ 2011/2012, 2016, ᐊᒻᒪᓗ 2021, ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᑦ ᖃᖓᑕᓪᓗᑎᒃ ᓇᓚᐅᑖᖅᓯᒪᔪᑦ ᐅᒥᓲᓂᕋᖅᑕᐅᓚᐅᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᖓᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᓯᑕᐃᐳᑕᓐ ᐊᓯᖏᓪᓗ. 2014, ᑎᒃ ᐊᓯᖏᓪᓗ. 2016, ᐊᑦᑭᓐᓴᓐ ᐊᓯᖏᓪᓗ. 2022) ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ (ᐊᑉᐹᑦ ᐊᓯᖏᓪᓗ. 2015, ᐊᑉᐹᑦ ᐊᓯᖏᓪᓗ. 2018, ᓄᐊᑦᑐᓚᑉ ᐊᓯᖏᓪᓗ. 2022). ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ, ᐅᓇ 2011 ᓇᓚᐅᑖᖅᓯᒪᓚᐅᖅᑐᖅ 1030 (CI = 754-1406), ᓄᖑᑉᐸᓪᓕᐊᔪᑦ ᐃᒪᓐᓇᐅᑎᒋᓕᖓ 18% ᑕᕝᕗᖓ 842 (CI = 562-1121) ᑕᐃᑲᓂ 2016 ᐊᒻᒪᓗ ᓱᓕᒃᑲᓐᓂᖅ 27% ᓄᖑᑉᐸᓪᓕᐊᔪᑦ ᐅᕗᖓ 618 (CI = 425-899) 2021-ᒥ. ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ, ᐅᓇ 2011/12 ᓇᓚᐅᑖᖅᓯᒪᓚᐅᖅᑐᖅ 943 (CI = 658-1350), ᓄᖑᑉᐸᓪᓕᐊᔪᑦ 17% ᑕᕝᕗᖓ 780 (CI = 590-1029) ᑕᐃᑲᓂ 2016 ᐊᒻᒪᓗ ᐅᓄᖅᓯᕙᓪᓕᐊᔪᑦ ᐃᒪᓐᓇᐅᑎᒋᔪᖅ 43% ᑕᕝᕗᖓ 1119 (CI = 860-1454) 2021-ᒥ. ᑕᐃᑲᓂ ᓈᓴᐃᔪᓂ, ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᓇᓂᔭᐅᓚᐅᖅᑐᑦ ᑲᓇᖕᓇᖓᓂ Nelson River ᑕᐅᕗᖓ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑭᒡᓕᐊ ᖃᓅᑎᒋᓚᐅᕐᓂᖓ ᒫᓐᓇᒎᑦ ᑭᖑᓂᑦᑎᓐᓂ ᓯᒡᔭᓂ ᓈᓴᖅᑕᐅᔪᑦ ᑕᐃᑲᓂ 1970 ᑎᑭᖦᖢᒍ 2011 (ᓴᖅᑭᔮᖅᑐᖅ 1, ᐅᖃᖅᓯᒪᔪᖅ ᓄᓇᖓᓄᑦ 2) (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 2004, ᓯᑕᐃᐳᑕᓐ ᐊᓯᖏᓪᓗ. 2014, ᑎᒃ ᐊᓯᖏᓪᓗ. 2017, ᐊᑦᑭᓐᓴᓐ ᐊᓯᖏᓪᓗ. 2022). ᑕᐃᒪᓐᓇᐅᒐᓗᐊᖅᑎᓪᓗᒍ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᑯᐊ ᓇᓄᐃᑦ ᐅᐸᒃᑕᐅᓯᒪᓗᑎᒃ ᐱᓕᕆᔭᐅᓯᒪᓐᖏᑦᑐᑦ, ᖃᐅᔨᒪᓐᖏᑦᑐᖅ ᐅᓄᖅᓯᖕᒪᖔᑕ ᓇᓄᐃᑦ ᕿᓚᒥᐅᔪᒥ ᐅᕝᕙᓘᓐᓃᑦ ᐊᑯᓂᐅᔪᒥ ᐊᓯᕈᕐᓇᐅᑎᓪᓗᒍ ᓇᒧᖕᒑᕗᖏᑦ ᓇᓕᐊᑐᐃᓐᓇᖅ ᐃᓚᖏᓪᓗᓐᓃᑦ ᐅᕝᕙᓘᓐᓃᑦ ᖃᓕᓅᓕᖅᓯᒪᕙᑦ ᓂᕐᔪᑎᑦ ᑕᒪᐃᓐᓂᑦ ᑖᒃᑯᓇᖓᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᒌᑦ ᓇᓄᖏᑦ. Resolving ᐋᖅᑭᒃᖢᒍᑦ ᐅᓇ ᖃᐅᔨᒪᓇᑦᑎᐊᖏᑦᑐᖅ ᐱᒻᒪᕆᐅᑐᓂ ᐊᐅᓚᑦᑎᔨᑦ ᑕᐅᑦᑐᐊᓂᓪᓕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᒪᕐᒥᒃ ᐃᓚᒌᑦ ᓇᓄᐃᑦ ᐊᖑᓇᓱᒃᑕᐅᖃᑦᑕᖅᑐᑦ, ᐊᒻᒪᓗ ᐊᕐᕌᒍᑕᒫᑦ ᑎᑭᐅᑎᔪᓐᓇᖅᑕᖏᑦ ᐊᖑᓇᓱᒋᐊᖃᕐᓂᖏᑦ ᓴᖅᑭᑕᐅᖃᑦᑕᖅᑐᑦ ᖃᓄᖅ ᐅᓄᖅᑎᒋᓂᖏᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᑦ ᒪᓕᒃᑐᑦ ᒫᓐᓇ ᐃᓕᓐᓃᑦ ᐅᓄᖅᓯᓂᖏᑦ ᑭᒡᓕᐊᓂ ᐊᒻᒪᓗ ᐃᓯᒪᒋᑦᑐᐃᓐᓇᖅᑐᖅ ᑖᒃᑯᐊ ᓇᓄᐃᑦ ᐃᓗᐊᓃᑦᑐᑦ ᑭᒡᓕᐊᑕ ᑖᒃᑯᓇᖓᑦ ᐃᓚᖓᓄᑦ ᐅᑯᐊᓗ ᑕᑯᔭᐅᓯᒪᔪᑐᐊᑦ. ᑐᑭᓯᓪᓗᒍ ᖃᓄᖅ

ᐊᓯᐳᕐᒪᖓᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᕿᓚᒥ ᐅᕝᕙᓘᓐᓃᑦ ᓅᕕᖕᒪᑕ ᐅᕝᕙᓘᓐᓃᑦ ᐊᓯᑦᔨᕈᑎᕗᖅ ᐅᓄᕐᓂᖏᓐᓂᒃ ᑕᐃᒪᓗ ᐱᒻᒪᕆᐅᖕᒪᑦ ᖃᐅᔨᒪᓗᒋᑦ ᓴᐳᓐᓂᐊᕐᓯᒍᑎᓄᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᔾᔪᑎᓄᑦ.

ᑐᑭᓕᐅᕈᑎᒋᕙᓪᓕᐊᓇᓯᓚᐅᖅᑕᕗᑦ ᑕᒪᑐᒨᓇ ᐃᓕᑦᑎᓇᓯᒃᖢᑕ ᐃᒪᓐᓇ 1) ᖃᐅᔨᑦᑎᐊᕐᓗᒋᑦ ᐃᓚᒋᒐᔭᖅᑕᖏᑦ ᓇᓄᐃᑦ ᑖᒃᑯᓇᖔᖅᑐᑐᖃᓂᑦ ᖃᓂᒋᔭᖏᓐᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ/ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓄᓇᐃᑦ ᑭᓪᓕᐊᓂ ᐊᒻᒪᓗ 2) ᖃᐅᔨᓇᓱᑦᑎᐊᕐᓗᒋᑦ ᐊᕙᑖᓂ-ᐊᕐᕌᒍᒥ ᐊᔾᔩᖏᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᓇᒧᖔᓯᒪᓂᖏᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᖅ ᓯᑯᐊ.

ᖃᐅᔨᓴᕈᓯᑦ

ᐊᑐᖅᓯᓚᐅᖅᑐᒍᑦ ᖃᐅᖓᓂᑦ ᐲᖅᓯᓪᓗᑕ ᐊᐅᖏᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᒋᓪᓗᒋᑦ ᑕᒫᓂᕐᕆᔪᓂᑦ ᓇᓄᕐᓂᑦ ᓯᓪᔭᒥ ᓄᓇᕗᑦ-ᒫᓂᑑᐸ ᑭᓪᓕᐊᓂ ᑕᐅᕗᖓ ᔭᐃᒻᔅ ᐸᐃ, ᐋᓐᑎᐅᕆᐅ (ᓴᕿᔮᖅᑐᖅ 1). ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ ᐱᖓᓱᓪᓗᐊᑕᓂᒃ ᑖᒃᑯᓇᖓᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᒌᑦ ᓇᓄᐃᑦ: ᓄᓇ 1, ᓄᓇᕗᑦ-ᒫᓂᑑᐸ ᑭᓪᓕᐊ ᑕᐅᕗᖓ Nelson River; ᓄᓇᖓ 2, Nelson River ᑲᓇᖕᓇᖓ ᑕᐅᕗᖓ ᐱᓇᖕᓇᖓᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓛᖏᑦ ᓇᓄᐃᑦ ᑭᓪᓕᐊᓂ; ᐊᒻᒪᓗ, ᓄᓇ 3, ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑭᓪᓕᐊ ᑕᐅᕗᖓ ᔭᐃᒻᔅ ᐸᐃ (ᓴᕿᔮᖅᑐᖅ 1). A 27.5 2017, ᑭᓛᒥᑐᔅ ᑭᓪᓕᓕᒫᖓ ᒫᓂᑑᐸ ᓯᓪᔭᖓ ᓴᕿᔮᖏᑦᑐᖅ ᐊᒃᑐᐃᒋᐊᖃᖏᒻᒪ ᖃᐅᔨᓴᕆᐊᒃᓴᕐᓗ ᓇᓄᕐᓂᒃ ᑕᑯᔭᖅᑐᖅᑕᐅᖃᑦᑕᕐᒪᑕ ᐳᓚᕋᑎᓂᑦ. ᓄᓇ 2 ᕿᓂᖅᑕᐅᓚᐅᖅᑐᖅ ᑕᒪᓐᓇᐃᓇᑐᐊᖅ ᑭᓪᓕᐊ ᓯᓪᔭᖓᓂ ᑕᐃᒪᖓᓂᑦ ᐊᕐᕌᒍᓂᑦ. ᑕᐃᑲᓂ 2017, ᓄᓇᒥ ᕿᓂᖅᑐᑦ ᖃᐅᔨᓂᐊᕐᒪᑕ ᑎᓯᖏᓐᓂᒃ ᓇᒥ. ᓄᓇᖓ ᐱᓕᕆᔭᐅᓚᐅᖅᑐᖅ ᐊᖏᓪᓕᒋᐊᕋᑕᕐᓂᐊᖅᑐᖅ; ᑕᐃᑲᓂ 2017 ᖃᐅᔨᓴᖅᑕᑐᐊᕆᓚᐅᖅᑕᕗᑦ ᒫᓂᑑᐸ, ᑕᐃᑲᓂ 2018 ᐊᒻᒪᓗ 2019 ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ ᐃᓗᑐᐊᖕᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑭᓪᓕᐊ, 2021 ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ 30 ᑭᓚᒦᑕᔅᒧ ᐱᓇᖕᓇᖓ Cape Henrietta Maria ᐊᒻᒪᓗ 2022 ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓯᓪᓚᓕᒫᖓᓂᒃ ᐋᓐᑎᐅᕆᔫᑦᑐᒥᒃ ᐃᓚᒋᔭᐅᓪᓗᓂ ᐱᓇᖕᓇᖓ ᐃᓚᖓ ᔭᐃᒻᔅ ᐸᐃ ᓯᓪᔭᖓ, ᐊᑭᔅᒥᔅᑭ ᐊᒻᒪᓗ Twin Islands.

ᖃᐅᔨᓴᕈᑎᑖᓚᐅᓐᖏᑦᑐᒍᑦ 2020 ᐱᔾᔪᑎᒋᓪᓗᒍ ᓄᕚᒃᔪᐊᕐᓇᖅ-19.

ᐃᓚᒌᑦ ᓇᓄᐃᑦ ᓇᑭᙶᖅᓯᒪᓂᑐᖃᖏᑦ

ᓄᐊᑦᑎᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᓴᕈᑎᒃᓴᓂᒃ ᓇᓄᕐᓂᑦ ᐱᔭᐅᔪᑦ ᐃᓚᒋᓪᓗᒍ ᐊᑯᓂᐅᔪᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᑦ ᐱᓕᕆᑎᑦᑎᔾᔪᑕᐅᔪᑦ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 1999, ᓚᓐ ᐊᓯᖏᓪᓗ. 2016) ᐅᕝᕙᓘᓐᓃᑦ ᐅᖓᓯᒋᔭᒥᑦ ᐊᑐᖅᑐᑦ ᓇᓕᐊᑐᐃᓐᓇᖅ 5cc ᑕᕆᐅᕐᒥᐅᑕᕐᓂᒃ ᓂᕿᖓᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᓂᖅ ᐅᕝᕙᓘᓐᓃᑦ 6cc ᓂᕿᖓᓂᑦ ᖃᐅᔨᓴᕈᑎᖅᑖᖅᖢᑎᒃ ᑲᐱᓪᓗᒍ (ᓄ ᑕᑦ ᒍᓕᐊᒻᔅᐳᐊᑦ, ᐱᓐᓯᕗᓂᐊ, ᔪᓇᐃᑎᑦ ᓯᑕᐃᑦᔅ) ᕐᑯᓕᒥᒎᓕᒃᑯᑦ ᕐᑯᑭᖅᑕᐅᓪᓗᑎᒃ (ᐸᒐᓄ ᐊᓯᖏᓪᓗ. 2014). ᑕᐃᒃᑯᓂ ᖃᐅᔨᓴᕈᑎᑖᕋᓱᒃᖢᑎᒃ, ᖃᐅᔨᓇᓱᕐᑯᔭᐅᓚᐅᖅᑐᒍᑦ ᐊᖑᑎ ᐊᕐᓇᕐᓘᓐᓃᑦ ᐊᒻᒪᓗ ᖃᓄᖅ ᐅᑭᐅᖃᖅᐸ ᐊᖏᓂᖓᒍᑦ ᐊᒻᒪᓗ ᓇᑭᖓᔭᐅᕙᑦ ᖃᓄᓇᖁᓂᖏᑦᑎᒍᑦ. ᐊᑐᓂ ᐋᕿᒃᓱᓚᐅᖅᑕᕗᑦ ᒪᓕᒃᖢᒋᑦ ᐊᑎᒃᑕᖅ ᐊᕐᕌᒍ ᐃᓗᐊᓂᑦ, ᐊᓈᓇᒥᓂᑦ ᐊᒫᒪᒃᑐᖅ, ᐃᓐᓇᖑᓯᔪᑦ (1-4 ᐅᑭᐅᓕᑦ), ᐊᒻᒪᓗ ᐃᓐᓇᐃᑦ (≥ 5 ᐅᑭᐅᓕᑦ).

ᐊᐅᑉ ᐃᓄᐊ ᐲᖅᓯᓚᐅᖅᑐᒍᑦ ᐊᑐᓂ ᓂᕿᖓᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᖅᖢᑕ ᐊᑐᖅᓯᓪᓗᑕ DNEASYᐊᐅᖏᖅᓯᔾᔪᑎ ᐊᒻᒪᓗ ᑕᖏᖓᓂᒃ ᐆᑕᖅᑎᑦᑎᔾᔪᑎ (ᖃᐃᔮᓐ, Hᐃᑕᓐ, ᔮᒪᓂ). (ᐊᑐᓂ ᖃᐅᔨᓴᕈᑎᑖᑦ ᑭᓯᖃᕐᒪᖔᑕ ᑕᐃᒃᑯᓂ 24 ᐊᐅᖏᖅᑎᑦᑎᒍᑦ ᖃᐅᔨᓴᕈᑎᑦ ᐃᓄᐊᒍᑦ ᑕᐃᒃᑯᓂ ᕐᑯᐱᕐᕈᕐᓂᒃ ᖃᐅᔨᓴᕐᕕᒃ ᐆᒪᔪᓂ ᐱᔨᑦᑎᕋᖅᑐᑦ (ᐃᓕᓐᓂᐊᕐᕕᒃᔪᐊᖓ ᐊᐃᐳᑕ, ᐃᒻᒪᑕ, ᐊᐃᐳᑕ). ᐊᐅᖏᖅᑎᑦᑎᒍᑦ ᒪᓕᒃᑕᐅᓂᖓ ᕐᑯᐱᕐᕈᖏᓐᓂᒃ ᐅᑯᐊᓗ; CXX20 ᐊᒻᒪᓗ CXX110 (ᐊᔅᑳᓐᑐ ᐊᓯᖏᓪᓗ. 1993) G1A, G10B, G1D, G10L (ᐸᐃᑦᑲᐅ ᐊᒻᒪᓗ ᓯᑐᓗᐱᒃ 1994) G10C, G10M, G10P, G10X (ᐸᐃᑦᑲᐅ ᐊᓯᖏᓪᓗ. 1995), UarMU05, UarMU10, UarMU23, UarMU26, UarMU50, UarMu51, UarMU59 (ᑕᐳᓕᑦ ᐊᓯᖏᓪᓗ. 1997), G10H, G10J, G10U (ᐸᐃᑦᑲᐅ ᐊᓯᖏᓪᓗ. 1998), MSUT-1, MSUT-2, MSUT-6, MSUT-8 (ᑭᑦᑕHᐃᐅᕋ ᐊᓯᖏᓪᓗ. 2000). ᐊᖏᓐᓇᖅᑐᖅ ᕿᖑᒥᒐᐅᑎ ᐊᖏᓪᓕᒋᐊᓘᖅ ᑎᓴᒪᐃᓪᓗᐊᖅᑐᖅ ᖃᓄᓇᑦᑐᑐᐃᓐᓇᓂᒃ ᓇᓗᓇᐃᖅᓯᓖᖅ ᐊᒻᒪᓗ ᐋᕿᒃᓯᔪᖅ 3 ᐃᓗᓪᓕᖅᓱᖅᓯᒪᖃᑎᓖᒃᖢᑎᒃ ᑖᒃᑯᐊ ᐃᒻᒥᖕᓄᑦ ᐊᑕᔪᑦ ABI 3730 ᑎᒥᐊᓄᑦ ᖃᐅᔨᓴᕈᑎ ᓈᓚᐅᑎᑦ ᐊᐅᖓᓂᑦ ᐃᓄᐊᒍᑦ ᖃᐅᔨᒪᔾᔪᑎᑦ ᐊᔾᔨᒌᙱᑦᑐᑦ (ᐃᓕᓐᓂᐊᕈᑎᑦ ᐆᒪᔪᓕᕆᓂᕐᒥᑦ, ᒍᐊᑦHᐊᒻ, ᒪᓯᑑᓯᑦᔅ, ᔪᓇᐃᑎᑦ ᓯᑕᐃᑦᔅ)). ᐊᖏᓂᖏᑦ ᐊᐅᑉ ᐃᓄᐊ ᑎᒥᐅᑉ ᐋᕿᒃᓯᐃᔭᖓᐊᑕᔪᑦ ᓈᓚᐅᑎᖏᑦ ᑎᒥᐊᓄᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᐃᓯᒪᓕᐊᖑᓚᐅᖅᑐᑦ Genescan 500 ᐊᖏᓂᖓ ᐱᐅᓯᖓᓂᒃ ᐊᒻᒪᓗ

ᐊᑐᓂ ᓇᓅᑉ ᐱᖓ ᐊᑎᖃᖅᑐᖅ ᐊᐅᖓᑕ ᐃᓄᐊ ᓇᓗᓇᐃᒃᑯᑕᕆᓪᓗᒍ 3.0 (ᐃᓕᓐᓂᐊᕈᑎᑦ ᑎᒥᐅᑉ ᐊᐅᓚᔾᔪᓯᖓᓂᒃ). ᑖᒃᑯᐊ ᐊᑐᓂ ᓇᓅᑉ ᐊᐅᖏᑦᑕ ᐃᓄᖏᑦ ᐊᑐᖅᑕᐅᓗᑎᒃ ᐲᖅᑕᐅᓚᐅᖅᑐᓂᑦ ᓂᕿᐊᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕆᔭᐅᓚᐅᖅᑐᑦ ᖃᓅᑎᒋᑦᑕᕐᒪᖔᑕ ᐊᑐᓂ ᓇᓄᖅ ᐊᑕᐅᓯᖅ ᐊᐅᖓ ᑕᐃᑲᖓᑦ ᑭᖑᓂᑦᑎᓐᓂ ᐊᑐᓂ ᖃᓄᕆᑑᓚᐅᕐᓂᖓ ᑕᒪᓐᓇ ᑲᔪᓯᓂᐊᖅᑐᖅ ᐊᑯᓂᐅᔫᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓗᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (3938) ᐊᒻᒪᓗ ᖃᓇᓴᐅᔭᒃᑯᑦ ᐊᑦᑕᑕᖅᓯᒪᔪᑦ ᐃᓕᑦᑎᓇᓱᒍᑎᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (864). ᐊᔾᔨᒃᑕᖅᑐᑦ ᐊᑐᓂ ᖃᐅᔨᓴᕈᑎᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ ᐊᐅᖏᑦᑎᒍᑦ ᐃᓄᖏᑦ ᒪᓕᒃᑕᐅᓪᓗᑎᒃ ᓴᓇᕈᑎᖏᑦ ᑲᑎᖓᔪᓂᑦ (ᐸᒃ 2001) ᐅᕝᕙᓘᓐᓃᑦ GenAIEx 6.5 (ᐱᒃᑳᓪ ᐊᒻᒪᓗ ᓯᒪᐅᔅ 2012) ᐊᔪᕐᓇᓚᐅᓐᖏᑦᑐᖅ ᑎᑭᐅᑎᓪᓗᒋᑦ 4ᓄᑦ ᐊᔾᔨᓐᖏᑦᑐᑦ ᐊᑐᓂᒧᑦ ᓇᓗᓇᐃᒃᑯᑕᑦ ᐊᐅᖏᑦᑎᒍᑦ. ᖃᓄᖅ ᐊᔾᔨᒃᑕᕐᓂᐊᕐᒪᖔᑕ ᓴᕿᔮᖅᑐᑦ ᖃᐅᔨᓇᓱᑦᑎᐊᑦᑎᐊᖅᓱᑎᒃ ᐊᒻᒪᓗ ᕿᔭᓈᖅᑕᐅᓪᓗᑎᒃ ᐊᔾᔨᖕᒪᒋᑦ ᑕᐃᒪᓗ ᐊᔾᔨᓐᖏᑕᖏᓐᓂᒃ ᐊᒥᓱᒃᑲᓐᓂᓂᓗ ᐅᖓᑖᓂ 4 ᐊᑐᓂ ᓇᓗᓇᐃᖅᑐᑦ ᐅᓄᓖᖑᓪᓗᑎᒃ ᐊᑕᐅᓯᐅᑦᑕᕐᓂᖏᑦ 2 loci. ᓇᓗᓇᐃᑦᑎᐊᓚᐅᖅᑕᕗᑦ ᓇᓄᐃᑦ ᐃᓚᖏᑦ ᑕᐃᑲᖕᖔᖅᓯᒪᔪᑦ ᐃᓚᒋᔭᐅᔪᑦ ᓯᕗᓪᓕᕐᒥᒃ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᒋᓚᐅᖅᑕᕗᑦ. ᐃᓂᓴᕆᔭᕗᑦ ᐅᓇ ᑎᓕᔭᐅᓯᒪᓂᖅᐳᑦ ᑕᐃᓖᑐᐃᓐᓇᖅᓯᒪᒐᔭᕋᓗᐊᖅᑎᓪᓗᒋᑦ, ᐱᓗᐊᖅᑐᒥᒃ ᖃᓂᒋᔭᖓᓃᑎᓪᓗᒋᑦ ᐃᓚᒋᑦᑕ ᑭᓪᓕᐊᓂ ᑭᓯᐊᓂ ᐃᒃᐱᒋᔭᐅᔪᖅ ᑐᓂᓯᓯᒪᔪᖅ ᑐᑭᓕᐅᕈᑎᒋᓂᐊᖅᑕᓄᑦ ᖃᓅᑎᒋᓇᔭᕐᒪᖔᑕ ᖃᖓᐅᓕᖅᐸᑦ.

ᐃᓗᐊᓂ ᐊᕐᕌᒍ ᐊᔾᔩᖃᑦᑕᓐᖏᑦᑐᑦ ᓇᒧᖔᓯᒪᕚᖏᑦ ᐊᒻᒪᓗ ᑕᕆᐅᖅ ᓯᑯᐊ ᖃᓄᐃᓕᖓᑎᑦᑎᓂᖓ

ᐊᑐᖅᑕᕗᑦ AMSR2 ᖃᖓᑦᑕᖅᑎᑕᐅᓯᒪᔪᑎᒍᑦ ᓴᕿᔮᖅᑐᑦ ᓇᔪᕐᕕᒋᖃᑦᑕᖅᑕᖏᑦ ᑕᕝᕗᓇ 3.5 x 3.5 ᑭᓛᑕᔅ (ᐃᓕᓐᓂᐊᕐᕕᒃᔪᐊᖅ ᐳᕆᒥᓐ, [Index of /amsr2/asi_daygrid_swath/n3125 (uni-bremen.de)] ᖃᐅᔨᓇᓱᑦᑎᐊᕐᓗᒍ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐊᔾᔨᒃᖃᑦᑕᓐᖏᑦᑐᑦ ᖃᓄᕆᑑᓂᖏᑦ ᓯᑯᐃᑦ. ᓇᓗᓇᐃᑦᑎᐊᕐᓗᒍ ᐊᒥᐊᒃᑯᕐᓗᐃᑦ ᓯᑯᐃᑦ ᑎᑭᐅᒪᔪᓐᓇᖅᑐᑦ 10% ᐅᑭᐅᒃᑯᑦ ᐊᖏᓘᖅ ᑕᒪᕐᒥᒃ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᓕᐅᑎᓯᒪᓇᓂ ᔭᐃᒻᔅ ᐸᐃ. ᓯᑯ ᓇᓗᓇᐃᑦᑎᐊᕐᓗᒍ ᐊᖏᒡᓕᒋᐊᕐᑎᑦᑎᕙᓕᒃ ᓴᕿᔮᕐᓂᖓ ᐊᔾᖑᐊᖓᒍᑦ ᑕᕆᐅᖅ ᓯᑯᖃᕐᓂᖏᑦ ≥ 30%. ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᕙᒃᓱᒍ ᐊᑐᖅᓯᓪᓗᓂ ᕿᑎᐸᓗᐊᓂᑦ ᑕᐃᑲᖓ ᓯᓕᓖᖓᓂᑦ ᓄᕗᐊᓂᑦ ᐃᒃᐱᐊᕐᔪᐊᑕ ᐊᒻᒪᓗ ᓯᑯᑦ ᐊᒥᐊᒃᑯᑦ ᓇᓗᓇᕈᓐᓃᖅᓱᑎᒃ, ᐊᖏᓐᓇᖅᑐᒃᑯᑦ ᐋᕿᒃᓱᐃᑎᓕᒃ, ᑲᔪᓯᓗᑎᒃ ᑲᓇᖕᓇᖓᓄᑦ (0) ᐅᕝᕙᓘᓐᓃᑦ ᐱᓇᖕᓇᖓᓄᑦ (1) ᐊᑐᖅᓯᓗᑕ ᐱᑕᖃᖅᑐᓂᒃ ᐊᖏᓂᖅᓴᐅᔪᓂ ᓄᓇᓂ ᑕᕆᐅᖅ ᓯᑯᐊᓂ ᐅᓪᓗᐊᒍᑦ ᐊᑐᖅᓯᓗᑕ ᓯᑯᓕᓛᖅᑲᓚᐅᕐᓂᖓ.

ᐅᓇ ᑲᓇᖕᓇᖕᒐ/ᐱᓇᖕᓇᖕᒐ ᓯᑯᖃᕐᓂᖕᒐ ᓇᓗᓇᐃᑦᑎᐊᓚᐅᖅᑐᖅ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑭᖑᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᕈᑎᑦ ᑕᑯᑎᑦᑎᓯᒪᔪᑦ ᓯᑯᖃᒪᓖᒃᑳᖕᒐᑦ ᑲᓇᖕᓇᖕᒐᓂ ᐃᒃᐱᐊᕐᔪᐊᑕ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᓯᒡᔭᖕᒐᓄᐊᖃᑦᑕᖅᑐᑦ ᐅᖕᒐᓯᖕᓂᖅᓴᓄᑦ ᑲᓇᖕᓇᖕᒐᓂ ᐊᔾᔨᒋᓐᖏᑕᖕᒐ ᑕᐃᒪᖕᒐᓂᑦ ᐊᕐᕌᒍᓂ ᓯᑯᒐᓖᖃᖅᓯᒪᓂᖕᒐᓂᑦ ᐱᓇᖕᓇᖕᒐᓂ (ᓯᑑᓕᖕ ᐊᓯᖕᒋᓪᓗ. 1999, ᓯᐅᓕ ᐊᓯᖕᒋᓪᓗ. 2013).

ᖃᐅᔨᓴᕐᓂᐊᖅᖢᒍ ᐃᓗᐊᓂ ᐊᕐᕌᒍ ᐊᔾᔨᒌᖃᑦᑕᓐᖏᓂᖕᒐ ᓇᓄᐃᑦ ᓇᒧᖕᖔᓯᒪᓂᖕᒋᑦ, ᐆᒃᑐᕋᖅᓯᒪᕙᒍᑦ ᓇᔪᖅᐸᒃᑕᑐᖃᕐᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᑯᐊᓗ ᓇᓗᓇᐃᑦᑎᐊᕐᓗᒍ ᑕᐃᒫᒃ ᐅᖕᒐᓯᖕᓂᖕᒐ ᐊᕙᑖᓂᑦ ᐊᑕᐅᓯᖅ ᓇᓄᖅ ᐃᓂᖕᒐ ᑕᐃᑲᓂᑦᑕᐃᓈᖃᑦᑕᖅᑐᖅ ᐊᕐᕌᒍᓂ ᐊᑐᖅᓯᓪᓗᑎᒃ ᐅᖕᒐᓯᒌᒃᑑᑎᒻᒪᕆᐋᓗᐃᑦ. ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᒥᓃᖕᒋᓂᖕᒋᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᑖᑦ 2017-ᒥᑦ 2019-ᒧᑦ, ᖃᐅᔨᓴᕈᓐᓇᖅᑕᑐᐊᕚᑦ ᓇᔪᖅᐸᒃᑕᑐᖃᕐᒥᖕᓃᓐᖏᑦᑐᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ. ᑕᐃᒪᓐᓇᐅᒐᓗᐊᖅᑎᓪᓗᒍ, ᐊᖏᔪᑦ ᖃᐅᔨᓴᕈᑎᑖᑦ 2021-ᒥ ᐊᒻᒪᓗ 2022-ᒥ, ᐊᔪᖏᑎᑦᑎᓚᐅᖅᑐᖅ ᐅᕙᑦᑎᓐᓂᒃ ᖃᐅᔨᓇᓱᑦᑎᐊᖅᖢᒋᑦ ᐊᓯᐊᓄᐊᖅᓯᒪᔪᑦ ᑕᒪᕐᒥᒃ ᐅᑭᐅᖏᑦᑎᒍᑦ ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐊᕐᓇᐃᓪᓗ. ᓇᔪᖅᐸᒃᑕᑐᖃᕐᒥᖕᓃᓐᖏᑦᑐᑦ ᓇᓗᓇᕈᓐᓃᖅᓯᔾᔪᑎᖃᓚᐅᖅᑐᑦ ᐊᑐᖅᓯᓪᓗᑎᒃ ᓈᓴᐅᓯᓇᔾᔪᑎᑦ ᖃᓄᖅ ᐋᖅᑭᒃᑕᑦᑕᕐᒪᖔᑕ ᑐᑭᓯᓂᐊᖅᖢᒍ ᐃᓕᖅᑯᓯᓇᕙᒃᑕᖏᑦ ᓇᒧᖕᒐᐅᕙᒍᓯᖏᑦ. ᐊᔾᔨᒌᖏᓂᖏᑦ ᖃᐅᔨᓴᓇᓱᒃᑕᐅᑦᑎᐊᓕᖅᖢᓂ ᐊᑐᖅᓯᓪᓗᑎᒃ ᐊᑕᐅᓯᕐᒧᐊᖕᒐᔪᖅ ANOVA. ᐊᔾᔨᒌᖏᓂᖏᑦ ᓇᔪᖅᐸᒃᑕᑐᖃᕐᒥᖕᓃᓐᖏᑦᑐᑦ ᐊᕙᑖᓂᑦ ᑖᒃᑯᐊ ᐃᓱᒪᓕᐅᓚᐅᖅᑐᑦ ᐊᑐᖅᓯᓗᑎᒃ ᑕᐃᑲᓃᑦᑐᓃᖅᑎᓪᓗᒋᑦ Tukey HSD ᖃᐅᔨᓴᕈᑎᖏᑦᑎᒍᑦ. ᖃᐅᔨᓇᓱᑦᑎᐊᖅᖢᒋᑦ ᑭᓱᒧᑦ ᑕᕆᐅᖅ ᓯᑯᐊ ᓄᒃᑕᖅᒪᖕᓖᑦ ᑕᐅᕗᖕᒐ ᐊᑐᖅᓯᓚᐅᖅᑐᑦ ᐃᓂᐊᓄᑦ ᐅᑎᖅᑕᑦᑕᕐᓂᖕᒐ ᑲᓇᖕᓇᖕᒐᓄᖕᒐᐅᕗᑦ ᓇᔪᖅᐸᒃᑕᑐᖃᓐᖏᑕᖕᒋᓐᓃᑦᑐᑦ (0) ᐅᕝᕙᓘᓐᓃᑦ ᐱᓇᖕᓇᖕᒐᓄᖕᒐᐅᕗᑦ ᓇᔪᖅᐸᒃᑕᑐᖃᓐᖏᑕᖕᒋᓐᓃᑦᑐᑦ (1) ᐊᑐᖅᓯᖕᒪᑕ ᐊᔾᔨᒋᓐᖏᖅᑐᓂᒃ ᐊᒻᒪᓗ ᓇᓅᓐᓂᖏᑦ ᓯᑯᓖᑦ ᑕᐃᒪᓗ ᓂᕿᒃᓴᖏᑦ ᐊᔾᔨᒋᓐᖏᒻᒪᑕ. ᖃᐅᔨᓴᕈᓯᑦ ᐊᖏᔪᑦ, ᐊᑐᖅᓯᓚᐅᖅᑐᑦ ᑕᒪᐃᑲᔭᕐᓂᖕᒐ - ᐊᖏᓂᖕᒐ ᖃᐅᔨᓴᖅᖢᒋᑦ ᑕᒪᑐᒧᖕᒐ ᑕᒪᐃᑲᔭᖏᓂᖏᑦ ᖃᐅᔨᒪᓪᓗᒋᑦ. ᑭᓯᐊᓂ ᐅᖃᖅᓯᒪᒍᓂ, ᑕᒪᕐᒥᒃ ᓈᓴᐅᓯᓇᓯᒪᔪᑦ ᖃᐅᔨᓴᕈᑎᖏᑦ ᐃᓱᒪᒋᔭᐅᑦᑎᐊᓚᐅᖅᑐᑦ ᐃᒪᓐᓇ ᐊᖏᕚᑎᒋᓂᖕᒐᓂᒃ P≤0.05.

ᑐᑭᓯᓇᖅᓯᔪᑦ ᖃᐅᔨᓴᕈᑎᓂᑦ

ᐱᓚᐅᖅᑐᒍᑦ 727 ᐊᐅᑉ ᐃᓄᐊ ᖃᐅᔨᓴᕈᑎᑖᓂᒃ ᑖᒃᑯᓇᖕᒐᑦ 458 ᐊᔾᔨᒋᓐᖏᑦᑐᑦ ᐊᑐᓃᖅᑐᑦ ᑕᐃᑲᖕᒐᑦ

ᓄᓇ 2 (ᓴᕿᔮᖅᑐᖅ 2) ᑕᐃᑲᓂ 2017-2022 ᐊᒻᒪᓗ 570 ᐊᐅᑉ ᐃᓄᐊ ᖃᐅᔨᓴᕈᑎᑖᓂᒃ ᑖᒃᑯᓇᖓᑦ 514 ᐊᑐᓃᖅᑐᑦ ᑕᐃᑲᖓᑦ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᓄᓇ 3) ᑕᐃᑲᓂ 2018-2022. ᐅᓇ ᐊᕐᕌᒍᒥ ᐳᓴᑎᖓ ᑭᖑᓂᑦᑎᓐᓂ ᓂᕕᖓᑖᓕᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᖃᓂᒋᔮᓂᑦ ᑕᒫᖓᑦ 30% ᑎᑭᓪᖢᒍ 43%. ᑕᐃᒪᖓᓂᓕᒫᖅ ᐊᕐᕌᒍᓂ, ᑕᒪᕐᒥᐸᓗᒃ ᓇᓄᐃᑦ ᑕᐃᑲᓃᑦᑐᑦ ᓄᓇ 2 ᐱᓕᕆᐊᖑᓚᐅᖅᑐᖅ ᑖᒃᑯᓄᖓ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᖃᓂᒋ.ᖓ 19-28%), ᑕᐃᑲᓂᓗ ᐳᓴᑎᖓ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᖃᓂᒋᔮᓂ 11 ᑎᑭᓪᓗᒍ 15% ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓚᐅᖅᑐᑦ (ᓇᓗᓇᐃᖅᓯᔪᖅ 1). ᑕᐃᑲᓂ ᓄᓇ 2, ᐱᓚᐅᖅᑐᒍᑦ 360 ᓂᕿᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᖅᖢᑕ ᑖᒃᑯᓇᖓᑦ 268 ᐊᑐᓂ ᓇᓄᕐᓂᑦ >1 ᐅᑭᐅᖃᖅᑐᖅ ᐊᕙᑖᓂ 2017 ᐊᒻᒪᓗ 2019 (ᓴᕿᔮᖅᑐᖅ 3). ᑖᒃᑯᓇᖓᑦ, 168 (63%) ᐊᔾᔨᒋᓚᐅᙱᑕᖏᑦ ᑭᖑᓂᑦᑎᓐᓂ ᓂᕕᖓᑖᓕᖕᓂᑦ ᓇᓄᑐᐃᓐᓇᓂᑦ, 72 (27%) ᐊᔾᔨᒌᓚᐅᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ, ᐊᒻᒪᓗ 28 (10%) ᐊᔾᔨᒋᓚᐅᖅᑕᖏᑦ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ (ᓲᓗ, 72% ᐊᒻᒪᓗ 28% ᑭᖑᓂᑦᑎᓐᓂ ᓂᕕᖓᑖᓕᑦ ᓇᓄᐃᑦ ᐅᑯᐊ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᐊᑐᓂ). ᓇᒧᖔᓯᒪᓂᖏᑦ ᑭᖑᓂᑦᑎᓐᓂ ᐱᔭᐅᕕᐅᓂᑯᑦ ᐃᓂᖏᑦ (n=221) ᑖᒃᑯᓄᖓ 100 ᓇᓄᐃᑦ ᑕᐃᑲᓃᑦᑐᑦ ᑕᒪᐃᓐᓂ ᓄᓇᖏᓐᓂ, ᑕᐃᒪᐃᓐᓇᐅᒐᓗᐊᖅᑎᓪᓗᒍ, 76% ᐅᕙᓃᓚᐅᖅᑐᑦ ᓄᓇ 1, 17% ᑕᐃᑲᓂ ᓄᓇ 3 ᐊᒻᒪᓗ 7% ᑕᐃᑲᓂ ᓄᓇ 2 (ᓴᕿᔮᖅᑐᖅ 4). ᕿᓕᕈᓕᑦ ᐃᑭᓕᑦ / ᐊᓐᓇᐃᔭᐅᓂᑯᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᖅ ᐊᕐᕌᒍᑕᒫᖅᖢᒋᑦ ᓄᓇ 1 ᑕᐃᒪᖓᓂᑦ 1980, ᑕᐃᒪᐃᓐᓇᐅᒐᓗᐊᖅᑎᓪᓗᒍ, 2009 ᑭᖑᓪᓕᖅᐹᓇᓚᐅᖅᑕᖓᑦ ᕿᓕᕈᓕᑦ ᐃᑭᓕᑦ / ᐊᓐᓇᐃᔭᐅᓂᑯᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᖅ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᖓᓂ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᕿᓕᕈᖅᑖᕐᓂᑯᑦ ᐊᕙᑖᓂ 2012 ᐊᒻᒪᓗ 2015 ᑕᒫᓂ ᔭᐃᒻᔅ ᐸᐃ ᓯᒻᔭᖓᓂ ᐊᒻᒪᓗ ᐊᑭᒻᒥᔅᑭ ᕿᑭᖅᑕᖅ). ᑖᒃᑯᓇᖔᖅᑐᑦ ᑭᖑᓂᐊᒍᑦ 2017-ᒥᑦ 2019-ᒧᑦ ᖃᐅᔨᓴᕈᑎᑖᑦ ᐃᓚᖃᓚᐅᖅᑐᑦ ᓇᓐᓄᒃᑕᐅᔪᑐᐊᓂᒃ ᐊᕙᑖᓂᑦ 2010 ᐋᕿᒍᑎᓚᐅᖅᑐᖅ 71 (26%) ᐊᑐᓂ ᐊᔾᔨᒋᔭᖏᑦ ᓂᕕᖓᑖᓕᖅᑕᐅᓚᐅᖅᑐᑦ ᓇᓄᐃᑦ ᐊᐅᖏᑦᑎᒍᑦ ᖃᐅᔨᓴᕈᑎᓃᑦᑐᑦ. ᑖᒃᑯᓇᖓᑦ, 43 (16%) ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ, ᐊᒻᒪᓗ 28 (10%) ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ (ᓲᕐᓗ 61% ᓂᕕᖓᑖᓕᖅ ᑖᒃᑯᓇᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ 39% ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ).

2021-ᒥ, ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ 140 ᓇᓄᐃᑦ ᐊᑐᓂ ᑕᐃᑲᓂ ᓄᓇ 2 >1 ᐊᕐᕌᒍᓕᑦ. ᑖᒃᑯᓚᖓᖅ, 96

(69%) ᐊᔾᔨᒋᓚᐅᖅᑕᖏᑦ ᓇᐃᑦ ᐊᐅᖏᑦᑎᒍᑦ ᖃᐅᔨᓴᖅᓯᒪᔭᖅᑎᓐᓂᑦ (ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᐅᕙᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᑦ ᑕᐃᑲᓂ 2017-ᒥᑦ 2019-ᒧᑦ; ᓴᕿᔮᖅᑐᖅ 5). 87 (62%) ᑖᒃᑯᐊ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᐊᒻᒪᓗ 9 (6%) ᑖᒃᑯᐊ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᑕᒪᕐᒥᒃ ᐱᐅᒃᑲᓐᓂᓯᒪᔪᕕᓃᑦ ᑎᑎᕋᖅᓯᒪᔪᑐᖃᑦ ᐃᒻᒪᑲᓪᓚᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ (ᓇᓗᓇᐃᖅᓯᔪᖅ 1, ᓴᕿᔮᖅᑐᖅ 6). 2021-ᒥ, ᓄᐊᑦᑎᓚᐅᕆᕗᒍᑦᑕᐅᖅ 183 ᖃᐅᔨᓴᕈᑎᑖᖅᓱᑕ (102 (55%) ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, 12 (7%) ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᖑᑎᑦ, 55 (30%) ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᐊᒻᒪᓗ 14 (8%) ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ) ᑖᒃᑯᓇᖓᑦ ᐊᑐᓃᖅᓱᒋᑦ ᓇᓄᐃᑦ >1 ᐅᑭᐅᖅᑕᖅᑐᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᒪᓗ ᑖᒃᑯᐊ 64 (35%) ᐊᔾᔨᒋᓚᐅᖅᑕᖏᑦ ᐊᑐᓂ ᐊᐅᖃᑎᒋᓪᓗᒋᑦ (ᓴᕿᔮᖅᑐᖅ 7) ᑖᒃᑯᓇᖓᑦ 34 (53%) ᐊᔾᔨᒋᓚᐅᖅᑕᖏᑦ ᓂᕕᖓᑖᓕᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᐊᒻᒪᓗ 30 (47%) ᐊᔾᔨᒋᓚᐅᖅᑕᖏᑦ ᓂᕕᖓᑖᖃᓐᖏᑦᑐᑦ ᓇᓄᐃᑦ ᑭᖑᓂᑦᑎᓐᓂ ᓂᕿᖏᓐᓂᒃ ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᓯᒪᔪᑦ (ᓴᕿᔮᖅᑐᖅ 8). ᑖᒃᑯᓇᖓᑦ 64 ᐊᔾᔨᒌᓚᐅᖅᑐᑦ, 45 (25%) ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᑕᐃᑲᙶᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᐊᒻᒪᓗ 19 (10%) ᑕᐃᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ. ᑖᒃᑯᓇᖓᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᕐᓂᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (78%) ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, 16% ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ, ᐊᒻᒪᓗ 6% ᐃᓐᓇᕈᓕᖅᑐᑦ. ᑖᒃᑯᓇᖓᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᔪᑦ ᑕᐃᑲᖓᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ, 90% ᑎᖑᓂᑦᑎᓐᓂ ᐱᔭᐅᒃᑲᓐᓂᕐᓂᑯᑦ ᑕᐃᙶᖅᑐᑦ ᓄᓇ 2.

2022-ᒥ ᖃᐅᔨᓴᓚᐅᖅᑐᒍᑦ ᐊᕙᒻᒧᑦ ᓄᒃᑕᖅᑐᓂᒃ ᓇᓄᐃᑦ ᐃᓚᖐᑦ ᕿᓂᖅᓱᑕ ᑭᖑᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᕈᑎᑖᕕᓂᑦᑎᓐᓂᒃ 2021-ᒥ. 2021-ᒥ ᓄᐊᑦᑎᓚᐅᖅᑐᒍᑦ ᓄᐊᑦᑎᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᓴᕈᑎᑖᓂᒃ ᑖᒃᑯᓇᖓᑦ 254 ᓇᓄᕐᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐅᑯᐊᓗ 120 ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ 2022-ᒥ; 111 (93%) ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᒃᑲᓐᓂᖅᓱᑎᒃ ᓇᓄᐃᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐊᒻᒪᓗ 9 (7%) ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᓴᕿᔮᖅᑐᖅ 9). ᖃᐅᔨᓴᕈᑎᑖᓚᐅᖅᑐᒍᑦ 209 (ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᐃᓅᓴᖅᑐᑦ ᐊᕐᕌᒍᒥ) ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᐅᑯᐊᓗ 72 ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ 2022-ᒥ 46 (64%) ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ 26 (36%) ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ (ᓴᕿᔮᖅᑐᖅ 10). ᑖᒃᑯᓇᖓᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᖑᔪᓂᑦ ᑕᐃᑲᖓᑦ ᓂᒋᐊᓂ

ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᐅᑯᐊᓗ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᑲᓂ 2022, 62% ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, 12% ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ, 12% ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᖑᑎᑦ ᐊᒻᒪᓗ 15% ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ. ᑕᒪᕐᒥᒃ ᐊᑐᓂ, ᐅᑯᐊ ᓈᓴᐅᑎᑦ ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᐱᑕᖃᕐᒪᑦ ᐃᒻᒪᖄ 29% ᓇᓄᖃᒃᑲᓐᓂᖅᑐᖅ ᐊᑭᓐᓇᖅᐸᓯᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᑲᓂ 2022 ᐊᔾᔨᒍᓇᓂᐅᒃ 2021-ᒥ. ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ ᐅᓄᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐅᑭᐅᖏᑦᑎᒍᑦ ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐊᕐᓇᐃᑦ ᐊᕕᒃᑐᖅᓯᒪᓪᓗᒋᑦ ᓈᓴᖅᑕᓂ ᓇᓄᐃᑦ ᓄᓇ 2 ᑕᐃᑲᓕ ᑕᒪᕐᒥ ᐊᕐᕌᒍᓂ ᐃᒫᖑᑎᒋᔪᖅ 77%, 69%, 67%, 57%, ᐊᒻᒪᓗ 62% ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᓂ 2017-ᒥᑦ 2019-ᒧᑦ, 2021, ᐊᒻᒪᓗ 2022, ᐊᑐᓂ.

ᐃᓗᐊᓂ-ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔩᖃᑦᑕᓐᖏᑦᑐᑦ ᓇᒧᖔᓯᒪᓂᖏᑦ ᐊᒻᒪᓗ ᑕᕆᐅᖅ ᓯᑯᐊᑕ ᖃᓄᕆᑑᓂᖏᑦ

ᐱᑕᖃᖅᑐᖅ ᐃᓗᐊᓂ-ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔩᖃᑦᑕᓐᖏᑦᑐᑦ ᑕᕆᐅᖅ ᓯᑯᖏᑦ ᖃᓄᕆᑑᓂᖏᑦ. ᑕᐃᑲᓂ 2017, 2019, 2020, ᐊᒻᒪᓗ 2022 ᓯᑯᒐᓛᑦ ᑕᐅᕙᓃᓚᐅᖅᑐᑦ ᐱᓇᖕᓇᖓᓂ, ᐃᒪᐃᒻᒪᑦ 2018 ᐊᒻᒪᓗ 2021 ᑲᓇᖕᓇᖓᓃᓪᖢᑎᒃ (ᓴᖂᔭᖅᑐᖅ 11). ᖃᑦᓯᐅᑦᑕᕐᓂᖏᑦ ᐊᕙᑖᓂᑦ ᐊᔾᔨᒌᓐᖏᑑᑕᐅᔪᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ ᓇᔪᖅᐸᒃᑕᑐᖃᒥᓃᓐᖏᑦᑐᑦ 82%-ᖑᓚᐅᖅᑐᑦ ᑕᐃᑲᓂ 2017/2018, 84% ᑕᐃᑲᓂ 2018/2019 ᐊᒻᒪᓗ 87% ᑕᐃᑲᓂ 2021/2022. ᑕᐃᑲᓂ 2017/2018 ᑐᑭᖃᖅᑐᖅ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᓯᓛᖑᓚᐅᖅᐳᖅ ($\overline{x}$=120 ᑭᓚᒦᑕᔅ, ᖃᓂᒋᔭᖓᓂ 0.04 – 387 ᑭᓚᒦᑕᔅ, n=32), ᒪᓕᒃᑕᐅᓪᓗᓂ 2021/22 ($\overline{x}$=104 ᑭᓚᒦᑕᔅ, ᖃᓂᒋᔭᖓᓂ 0.07 – 411 ᑭᓚᒦᑕᔅ, n=113) ᑖᓐᓇᓗ 2018/2019 ᖃᓂᓛᒧᑦ ($\overline{x}$=71 ᑭᓚᒦᑕᔅ, ᖃᓂᒋᔭᖓᓂ 0.2 -260 ᑭᓚᒦᑕᔅ, n=80) (ᓴᖂᔭᖅᑐᖅ 12). ᐊᖏᔪᒻᒪᕆᐋᓗᒃ ᐊᔾᔨᒌᓐᖏᓂᖓ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᓯᒋᔭᖓᓄᐊᖅᑐᑦ ᐃᓐᓇᕐᓄᑦ ᐊᖑᑎᓄᒃ (ᐊᑕᐅᑦᑎᒧᑦ, ANOVA, $F_{2, 222}$ =4.175, P=0.02). ᐊᖏᔪᒻᒪᕆᐋᓗᒃ ᐊᔾᔨᒌᓐᖏᓂᖓ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᓇᓂᔭᐅᓚᐅᖅᑐᑦ ᑕᐃᑯᖓ 2017/2018 ᐊᔾᔨᒋᓇᓂᐅᒃ 2018/2019-ᒧᑦ (ᐊᓂᒍᖅᓯᒪᓕᖅᑎᓪᓗᒍ ᑐᑭᐅᑉ HSD ᖃᐅᔨᓴᖅᑕᖓ =0.03) ᐊᒻᒪᓗ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐊᕙᑖᓂ 2018/2019 ᐊᒻᒪᓗ 2021/2022 ᓴᖂᔭᑦᑎᐊᖅᑐᖅ ᐊᔾᔨᒌᓐᖏᓂᖓ (P=0.06).

ᐊᖏᔪᒻᒪᕆᐋᓗᒃ ᐊᔾᔨᒌᓐᖏᓂᖓ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᓯᒋᔭᖓᓄᐊᖅᑐᑦ ᐊᕙᑖᓂᑦ ᐅᑭᐅᖃᑎᒌᓐᖏᑦᑐᑦ ($F_{3, 185}$=3.57, P=0.02). ᖃᑦᓯᐅᑦᑕᕐᓂᖏᑦ ᐊᕙᑖᓂᑦ ᐊᔾᔨᒌᓐᖏᑑᑕᐅᔪᑦ

ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐊᑐᓂ ᐅᑭᐅᓕᑦ 2021/22 87%-ᖑᓛᐅᖅᑐᖅ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, 66% ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᐊᒻᒪᓗ 77% ᑕᒪᕐᒥᒃ ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ ᐊᒻᒪᓗ ᐊᖑᑎᑦ. ᐅᓇ ᑐᑭᖃᖅᑐᖅ ᓇᔪᖅᐸᒃᑕᓐᖏᓃᓐᖏᑦᑐᑦ ᐅᓄᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ, ($\overline{x}$=184ᑭᓚᒦᑕᔅ, ᖃᓂᒋᔭᖓᓂ 15-438 ᑭᓚᒦᑕᔅ, n=20), ᑖᒃᑯᐊᓗ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ (ᐅᖃᐅᓯᐅᔪᑦ ᖁᓛᓂ) ᑖᒃᑯᐊᖑᓕᕆᓪᓗᑎᒃ ᐃᓐᓇᕈᖅᐸᓪᓕᐊᔪᑦ ᐊᖑᑎᑦ ($\overline{x}$=100 ᑭᓚᒦᑕᔅ,ᖃᓂᒋᔭᖓᓂ 0.27-332 ᑭᓚᒦᑕᔅ, n=29) ᐊᒻᒪᓗ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ($\overline{x}$=80ᑭᓚᒦᑕᔅ, ᖃᓂᒋᔭᖓᓂ 2.7 - 195 ᑭᓚᒦᑕᔅ, n=27) (ᓴᕿᔮᖅᑐᖅ 12). ᐊᓂᒍᖅᓯᒪᓕᖅᑎᓪᓗᒍ ᑑᑭ HSD ᖃᐅᔨᓴᖅᓯᒪᔭᖏᑦ ᑕᑯᑎᑦᑎᓯᒪᔪᑦ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᖑᑎᑦ ᑖᒃᑯᓄᖕᖓ ᐊᔾᔨᒋᓐᖏᑕᖏᑦ ᐊᕐᓇᐃᑦ ᐊᖏᓂᖅᓴᒥᒃ ᐅᖓᓯᖕᓂᖅᓴᐅᔪᑦ ᑖᒃᑯᓇᖓᑦ ᐃᓐᓇᕐᓂᑦ ᐊᕐᓇᓂᑦ (P=0.02) ᐊᒻᒪᓗ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ (P=0.01) ᑭᓯᐊᓂ ᐊᔾᔨᒋᓐᖏᑦᑎᐊᖅᑕᖓ ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᖑᑎᑦ (P=0.07).

ᑭᓯᑐᐃᓐᓇᐃᑦ ᖃᓄᐃᓕᖓᓂᖏᓐᓄᑦ ᐃᓅᓯᕆᕙᒃᑕᖏᑦ ᒪᓕᒃᖢᒋᑦ ᐃᓚᖃᖅᑐᖅ ᓯᑯᒐᓛᑦ ᑕᐃᒪᓗ ᓂᕿᒋᓂᐊᖅᑕᖏᓐᓄᑦ ᐱᐅᓂᖅᓴᐅᖕᒪᑦ ᐱᔾᔪᑎᖃᕋᔭᓚᐅᓐᖏᑦᑐᒥᒃ ᑕᐃᒪᐃᓕᖓᔾᔪᑎᖏᑦ (ᖃᑦᓯᐅᑦᑕᕐᓂᖏᑦ ᐊᕙᑖᓂᑦ ᐊᔾᔩᓐᖏᑑᑕᐅᓂᖏᑦ ᖃᐅᔨᓴᖅᖢᒋᑦ, X^2= 20.80, df = 1, P<0.001). ᐅᓇ ᓂᕿᒋᓂᐊᖅᑕᖏᓐᓄᑦ ᑕᐃᒪᐃᒻᒪᑕᖃᐃᓚᖓᓂᖅ ᓇᔪᖅᐸᑦᑕᓐᖏᓐᓃᔾᔪᑎᔪᑦ = -1.0033 + (1.65115)*ᓯᑯᒐᓛᑦ. ᑐᑭᓯᓇᖅᓯᑎᓯᒪᔪᖅ ᖃᓅᑦᑕᕐᓂᖏᑦᑎᒍᑦ, ᑕᐃᒪᐃᒐᔭᕐᓂᖓᒍᑦ ᑖᒃᑯᐊ ᓇᓄᐃᑦ ᓇᒧᖕᒑᒐᔭᖅᑕᖏᑦ ᓄᒃᑕᖅᖢᑎᒃ ᐱᓇᖕᓇᖓᓄᑦ ᒪᓕᒃᑐᑦ ᓯᑯᒐᓛᑎᒍᑦ ᐱᓇᖕᓇᖓᑕ ᐃᓚᖓᓂ ᐃᒃᐱᐊᕐᔪᐊᓂ ᐃᒪᓐᓇᐅᓚᐅᖅᑐᖅ 5.01x ᐅᓄᕐᓂᖅᓴᑦ (95%CI 2.46 ᑕᕝᕙᖓ 10.92) ᖃᓃᑎᒋᓂᖓ ᓅᑦᑐᓄᑦ ᑲᓇᖕᓇᖓᓄᑦ.

ᐅᖃᖃᑎᒌᒍᑎᓂᑦ

ᑐᑭᓯᓗᒋᑦ ᖃᓄᖅ ᐊᔾᔩᖃᑦᑕᓐᖏᓂᖏᑦ ᑕᕆᐅᖅ ᓯᑯᖏᑦ ᐊᒃᑐᐃᔪᑦ ᓇᒧᖕᒑᖃᑦᑕᕐᓂᖏᓐᓂᒃ ᐊᒻᒪᓗ ᐅᓄᕐᓂᖏᓐᓂᒃ ᓂᕐᔪᑎᑦ ᐆᒪᔪᑦ ᐱᒻᒪᕆᐅᔪᑦ ᓴᐳᒻᒥᒋᐊᓕᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᔾᔪᑎᖃᕐᓗᑎᒃ ᓯᓚᐅᑉ ᐅᕐᖠᓯᕙᓪᓕᐊᓂᖓᓄᑦ ᐊᓯᕈᖅᑎᑦᑎᔪᑦ ᑲᔪᓯᔪᒥᒃ ᐊᑐᖅᓯᑎᑦᑎᕙᑦ ᐊᕙᑎᑦᑎᓂᒃ ᐊᓯᕈᖅᑎᑦᑎᓪᓗᑎᒃ ᑕᐃᑲᓂ ᓂᕆᐅᓐᖏᑦᑐᒃᑐᑦ ᓱᑲᓕᔾᔪᑦ (ᒫᓯᐊᔅ-ᕚᐅᕆᐊ ᐊᒻᒪᓗ ᐳᐅᔅᑦ 2018, ᐅᓚᓐᑕᓃ ᐊᓯᖏᓪᓗ. 2022). ᓱᓕ, ᑐᑭᓯᓗᒍ ᓴᖑᓂᖏᑦ ᓇᒧᖕᒑᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪᓗ ᐅᓄᕐᓂᖏᑦ

ᐱᒻᒪᕆᐅᔪᒥᒃ ᑕᐃᒪᐃᓕᖓᓕᖅᑐᑦ ᓄᓇᓕᖕᓄᑦ ᐅᑯᐊᓗ ᐊᑐᖅᓯᓲᖑᖕᒪᑕ ᐅᑭᐅᖅᑕᖅᑐᖅ ᐆᒪᔪᖏᓐᓂᒃ ᓂᕿᒃᓴᕆᖦᖢᒋᑦ ᐊᖑᓇᓱᖃᑦᑕᖅᖢᒋᑦ. ᑕᒪᑐᒪᓂ ᐃᓕᑕᐅᓇᓱᒃᑐᒥ, ᓇᓗᓇᐃᖅᓯᕗᒍᑦ ᐃᓚᒋᔭᐅᔪᑦ ᑖᒃᑯᓇᖖᖅᓯᒪᔪᑦ ᓇᓄᕐᓂᑦ ᓯᒡᔭᖓᓂ ᖃᓂᒋᔭᖓᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᑕ ᑕᓯᐅᔭᕐᔪᐊᖅ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓄᓇᐃᑦ ᐃᓪᓕᐊᓃᑦᑐᑦ, ᐊᑕᔪᑦ ᓇᒧᖖᓯᒪᓂᖏᑦ ᐊᓯᖑᖃᑦᑕᖅᑐᑦ ᑕᒪᐃᓐᓂ ᑭᓪᓕᐊᓂ ᑕᕆᐅᖅ ᓯᑯᐊ ᑕᐃᒪᐃᓕᖓᓂᖓᓄᑦ, ᐊᒻᒪᓗ ᓇᓗᓇᐃᖅᓯᒪᔪᒍᑦ ᐊᔾᔨᒌᙱᑦᑐᑦ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᙱᑦᑐᑦ ᐊᕙᑖᓂᑦ ᐅᑭᐅᖅᑲᑎᓖᙱᑦᑐᑦ.

ᑕᒪᕐᒥᐸᓗᒃ ᓇᓄᐃᑦ ᑕᐃᑲᓃᑦᑐᑦ ᓄᓇ 2 2017-ᒥᑦ 2022-ᒧᑦ ᑭᖑᓂᑦᑎᓐᓂᑦ ᐱᓕᕆᔭᐅᓚᐅᖅᓯᒪᙱᑦᑐᑦ ᐃᓚᒋᖦᖢᒍ ᖃᐅᔨᓴᕈᑎᑦᑕ ᐊᒻᒪᓗ ᒥᐊᓂᕆᔾᔪᑎᑦᑕ ᐱᓕᕆᑎᑦᑎᔾᔪᑕᐅᔪᑦ. ᐅᑯᐊ ᑭᖑᓂᑦᑎᓐᓂ ᐱᓕᕆᔭᐅᓯᒪᔪᑦ ᐱᔭᐅᖃᑦᑕᖅᖢᑎᒃ ᓇᓄᐃᑦ ᑕᒪᐃᓐᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᓖᑦ ᓄᓇᐃᑦ. ᑕᐃᑲᓂ 2017-ᒥᑦ 2019-ᒧᑦ, 28% ᓂᕕᖓᑖᓖᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓚᐅᖅᑐᑦ ᑕᐃᑲᖓᑦ ᓄᓇ 2 ᑖᒃᑯᐊ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ (39% ᑕᐃᒪᓗ ᓈᓴᖅᓯᒪᔪᖃᕐᕕᒃ ᐊᔪᕐᓇᓚᐅᖅᑐᑦ 2009-ᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑭᖑᓂᐊᓂ, ᐅᑯᐊᓗ ᑭᒡᓚᖅᑐᖅᑐᑦ ᐊᕐᕌᒍᓂ ᑕᐃᑲᖓᑦ ᓇᓄᐃᑦ ᓂᕕᖓᑖᖃᓕᖅᖢᑎᒃ ᑕᐃᑲᓂ ᓄᓇ 3) ᐊᔾᔨᒋᙱᑕᖓ 11%-ᑐᐃᓐᓇᖅ ᕿᑎᐊᓂ 1990ᓂ (ᓚᓐ ᐊᓯᖏᖦᖢ. 1997). ᐊᒥᓲᓂᖅᓴᐃᑦ ᑖᔅᓱᒪ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᖅ ᑕᐃᑲᓂ ᓄᓇ 2, ᑲᑎᖦᖢᒋᑦ ᐊᒥᓲᓂᖅᓴᐅᖦᖢᑎᒃ ᓂᕕᖓᑖᖃᖏᑦᑐᑦ ᓇᓄᐃᑦ ᑕᐃᑲᓂ, ᐃᒻᒪᖄᖅ ᐅᓄᖅᓯᓂᖏᓐᓄᑦ ᐃᓚᒋᑦ ᐱᕈᖅᐸᒍᓯᖏᑦᑎᒍᑦ, ᐅᓄᖅᓯᔪᑦ ᑭᓚᒻᒥᐅᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ/ᐅᕝᕙᓘᓐᓃᑦ ᓄᒃᑕᖅᐸᒃᑐᑦ ᑕᐃᑯᖓᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐃᓚᖏᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᑕᒪᕐᒥᒃ. ᐅᑯᐊ ᐃᑲᔪᖅᑕᐅᓂᖅᓴᐅᒐᒥᒃ ᓯᒡᔭᓂᑦ ᐅᓄᕐᓂᖅᓴᐅᕗᑦ ᑕᐃᑲᓂ ᓄᓇ 2 (ᓯᑕᐃᐳᑕᓐ ᐊᓯᖏᖦᖢ. 2014) ᐊᒻᒪᓗ ᓄᒃᑕᖅᑐᑦ ᐱᔾᔪᑎᒋᖦᖢᒍ ᐅᓄᕐᓂᖅᓴᐅᓕᖅᑎᖦᖢᒋᑦ ᓇᓚᐅᑖᖅᓯᒪᔪᑦ ᑖᒃᑯᓄᖓ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᓄᐊᑦᑐᓚᑉ ᐊᓯᖏᖦᖢ. 2022).

ᐊᑦᑎᑑᔪᖅ ᑭᖑᓂᑦᑎᓐᓂᑦ ᓂᕕᖓᑖᓕᖕᓂᑦ ᓇᓄᕐᓂᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᐱᔾᔪᑎᖃᖅᑐᒃᑕᐅᔪᑦ 10-ᐊᕐᕌᒍᓄᑦ ᐱᑕᖃᓚᐅᙱᑦᑐᑦ ᖃᐅᔨᓴᕈᑎᑦ ᑕᒪᑐᒪᓂ ᓄᓇᒥ ᑲᑎᖦᖢᒋᑦ ᐅᓄᖅᑑᒐᓗᐊᖅᑎᖦᖢᒋᑦ ᑖᒫᓂ ᐊᕕᒃᑐᖅᓯᒪᔪᓂ. ᑕᐃᑲᓂ 2017, ᐃᓚᖏᑦ ᐊᒥᓱᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᐅᖓᑖᓃᓚᐅᖅᑐᑦ ᐃᓕᓐᓇᓱᒃᑕᑎᓐᓂᑦ, ᑕᐃᒪᓐᓇᐅᒐᓗᐊᖅᑎᖦᖢᒍ ᖃᐅᔨᓴᕈᑎᑖᕗᑦ ᐊᖏᓂᖏᖏᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᓐᓂᒃ ᐊᑦᑎᑑᔪᖅ. ᓚᓐ ᐊᓯᖏᖦᖢ. (1997) ᖃᐅᔨᓴᕈᑎᑖᑦ ᐊᔾᔨᐸᓗᖏᑦ ᓄᓇᖓᓂ (ᓲᕐᓗ ᒫᓂᑑᐸᑐᐊᖅ ᐃᓚᖓ ᓄᓇ 2) ᐊᒻᒪᓗ ᐃᓚᖓᓂ

ᓂᒋᐊᓂ ᑕᓯᐅ.ᕐᔪᐊᖅ ᓇᓂᖕᒋᑦ ᐅᓄᕐᓂᖅᓴᐃᑦ 2017 ᓴᓇᔭᑦᑎᓐᓂᑦ. ᑕᐃᒪᐃᓚᐅᖅᑐᖅ, ᑕᐃᑲᓂ 2017, ᓯᑯᒐᓛᑦ ᐱᑕᖅᖃᑦᑕᓲᑦ ᐱᓇᖕᓇᖅᐸᓯᖕᒐᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᑕᐃᒪᓗ ᑎᓕᔭᐅᓯᒪᔪᒍᑦ ᓇᓄᐃᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖕᒋᑦ ᑕᐃᒪᐃᑎᑕᐅᔪᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓪᓗᑎᒃ ᐅᑯᐊᓗ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᓇᓄᐃᑦ ᐊᕐᕌᒍᑕᓖᑦ ᑕᐃᒪᖕᒐᓂᑦ ᐊᓂᒍᖅᑐᑦ 40, ᓄᓇ 2 ᐅᓄᖅᑑᓂᖅᓴᐅᔪᑦ ᓇᒋᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖕᒋᓐᓂᑦ ᑕᐃᒪᖕᒐᓂᑦ ᐊᓂᒍᖅᑐᓂ ᐊᕐᕌᒍᓂ ᓂᕕᖕᒐᑖᓕᖅᓯᒪᓐᖕᒋᒃᑲᓗᐊᕈᑎᑦ ᑕᐃᒪᖕᒐᓂᑦ 2009.

2021 ᐊᒻᒪᓗ 2022, ᑐᑭᓯᓇᖅᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᓛᑦ ᑕᒪᐃᓐᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᐊᒥᓱᓂᒃ ᐊᕐᕌᒍᓂ, 25-36% ᓇᓄᐃᑦ ᖃᐅᔨᓴᕈᖅᑖᕐᕙᐅᓯᒪᔪᑦ ᐊᑕᐅᓯᕐᒥ ᐃᓚᒌᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕙᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ ᐊᕐᕌᒍᖑᕐᒪᑦ ᖃᓂᒋᔭᖕᒋᓐᓂᑦ ᐃᓚᒌᑦ ᓇᓄᐃᑦ, ᓇᑐᓇᐃᖅᓯᕗᖅ ᐊᖕᒋᔪᒥᒃ ᐃᓚᒌᖕᓄᑦ ᓇᓄᕐᓄᑦ ᖃᓕᓅᓕᖅᓯᓯᒪᔪᑦ/ᑭᐳᖃᑦᑕᖅᑐᑦ ᖃᓂᒌᔭᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑭᒡᓕᐊ. ᑕᐃᒪᓗ ᐅᑯᐊ ᑐᑭᓯᓇᖅᓯᔪᑦ ᐊᒻᒪᓗ ᐊᓯᖕᒋᓐᓂᑦ, ᐃᓱᒪᓕᖅᓯᓯᒪᔪᑦ ᑖᒃᑯᐊ ᐊᒥᓲᓕᖅᓴᐅᖕᒪᑕ ᑭᐳᖃᑦᑕᖅᑐᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᕐᕋᒍ ᐃᓗᐊᓂ ᐊᔾᔨᒋᖃᑕᓐᖏᒻᒪᒍ ᓇᒧᖕᒌᓂᖕᒋᑦ ᓯᑯᒐᓛᑦ (ᓯᒃᑳᑦ ᐊᒻᒪᓗ ᓛᓯ 2010, ᑕᐅᓐᔅ ᐊᓯᖕᒋᓪᓗ. 2010, ᓯᐅᓕ ᐊᓯᖕᒋᓪᓗ. 2013, ᐅᓇ ᐃᓕᑕᐅᓇᓱᒃᑐᖅ), ᐱᓯᒪᓂᖃᕈᓐᓇᕐᒥᔪᖅ ᐊᓯᕈᖅᐸᓪᓕᐊᔪᑦ ᓇᒦᖃᑦᑕᓕᕐᓂᖕᒋᑦᑎᒍᑦ (ᓚᓐ ᐊᓯᖕᒋᓪᓗ. 2016, ᐊᑭᓐᓴᓐ ᐊᓯᖕᒋᓪᓗ. 2022, ᓄᐊᑦᑐᓚᑉ ᐊᓯᖕᒋᓪᓗ. 2022). ᑕᒪᓐᓇᐅᓚᓗᐊᕈᓂ ᑐᑭᓯᓇᖅᓯᔪᑦ ᖃᐅᔨᓴᕈᑎᓂᑦ ᐃᓱᒪᓕᖅᓯᓯᒪᔪᖅ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓄᒃᑕᖃᑦᑕᖅᑐᑦ ᐊᕙᑖᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓄᓇ 2 ᐊᓐᓂᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ, ᖃᑦᓯᐅᓂᖕᒋᑦ ᓯᕗᓪᓕᖅᐹᒥᒃ ᐱᔭᐅᔪᑦ ᑕᐃᑲᖕᒐᑦ ᓄᓇ 2 ᐅᓄᕈᓐᓃᖅᑐᑦ 23% ᐃᓱᓕᑎᓪᓗᒍ ᑕᒪᓐᓇ ᐃᓕᓐᓇᓱᒃᑕᕗᖅ, ᐃᓱᒪᓕᖅᓯᓯᒪᔪᑦ ᖃᓅᑎᒋᓂᖕᒐ ᐅᒃᐱᓇᔭᐅᒍᓂ ᑕᕝᕗᖕᒐ ᓄᓇ 2. ᐃᓱᒪᒋᓪᓗᒋᑦ ᑕᒪᕐᒥᒃ ᓇᓄᐃᑦ ᓂᕕᖕᒐᑖᖃᓐᖕᒋᑦᑐᑦ ᓄᓇ 2, ᐊᒥᓲᓂᖕᒋᑦ ᓇᓄᐃᑦ ᐅᓄᖅᓯᔪᑦ ᑕᐃᑲᓂ ᓄᓇᖕᒐᓂ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᓄᖅᓯᓂᖕᒐ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ, ᐃᓱᒪᓕᖅᓯᒪᕗᖅ ᐊᒥᓲᓂᖅᓴᐅᔪᑦ ᓇᓄᐃᑦ ᑕᕝᕙᓂ ᓄᓇ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓄᒃᑕᖅᑐᓂᑦ ᑎᑭᖃᑦᑕᖅᑐᓂᑦ ᑕᐅᕙᖕᒐᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᔅᓱᒪᓂ ᑕᕝᕙᖕᒐᑦ ᓄᓇ 1 (ᓯᑕᐃᐳᑕᓐ ᐊᓯᖕᒋᓪᓗ. 2014, ᓄᐊᑦᑐᓚᑉ ᐊᓯᖕᒋᓪᓗ. 2022). ᐃᑲᔪᖅᓯᒃᑲᓐᓂᕐᓗᒍ ᑕᒪᓐᓇ ᐅᒃᐱᓇᔭᖅ, ᓄᐊᑦᑎᓯᒪᔪᒍᑦ ᑖᒃᑯᓇᖕᒐᑦ ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᐊᕐᕌᒍᑕᓖᑦ 2018-ᒥᑦ 2021-ᒧᑦ ᑕᑯᔭᐅᓯᒪᔪᑦ ᖃᖕᒐᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓪᓗᑎᒃ ᓯᒡᔭᖕᒐᓂ York Factory ᑕᐅᕗᖕᒐ Fort Severn (ᐅᑯᐊᓗ ᐊᖕᒋᓂᖅᓴᐅᔪᑦ ᓄᓇᖕᒐ) ᑕᐃᑲᖕᒐᑦ 1972 ᑎᑭᓪᓗᒍ 1996

(ᓯᑑᓕᓐ ᐊᓯᖏᓪᓗ. 2004) ᐃᓱᒪᓕᒃᑲᓐᓂᕈᓐᓇᖅᑐᒍᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᔪᑦ ᓇᓄᐃᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ. ᖁᓛᓂ ᑕᑯᔭᑦᑎᓐᓂᑦ ᐃᓱᒪᓕᖅᓯᒪᕗᒍᑦ ᐅᑯᐊ ᖃᑦᓯᐅᓂᖏᑦ ᓇᓄᐃᑦ ᐅᓄᖅᓯᒋᐊᖅᑐᑦ ᖃᓄᖅ ᐊᔾᔨᒌᓐᖏᒪᖔ ᑕᐃᑯᖓ ᓄᓇ 1 ᑐᑭᓯᓇᕋᓂ. ᐅᑯᐊ ᓇᓂᔭᐅᔪᑦ ᐱᒻᒪᕆᐅᔪᒥᒃ ᑕᐃᒪᐃᑎᑦᑎᔪᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᕐᒪᖔᑕ ᐊᐅᓚᑕᐅᓗᑎᒃ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓄᓇ 2 ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐅᓄᖅᓯᕙᓪᓕᐊᖕᒪᑕ ᓇᒧᑐᐃᓐᓈᖅᐸᓪᓕᐊᔪᑦ, ᑕᐃᒪᓗ ᐅᓄᓗᐊᓕᖅᐸᑕ ᓇᓄᐃᑦ ᐃᒻᒪᖄ ᓄᒃᑕᕈᓐᓇᖅᑐᑦ ᐊᕙᑖᓂᑦ ᐃᓚᒋᓐᓂᑦ ᐊᕐᕌᒍᑕᒫᑦ. ᓱᓕ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᓄᖅᓯᓂᖏᑦ ᓈᓚᐅᑖᖅᓯᒪᔪᑦ ᑕᒪᐃᓐᓄᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᒫᓐᓇ ᐱᔭᐅᓯᒪᔪᓂᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᓂᑦ ᑕᐃᒪᓗ ᐃᓱᒪᓇᖅᑐᖅ ᑖᒃᑯᐊ ᑕᒪᕐᒥᒃ ᓇᓄᐃᑦ ᓈᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᐃᓚᒦᓄᑦ ᐅᑯᐊᓗ ᑕᑯᔭᐅᓯᒪᔪᑦ, ᕿᓚᒥ ᐅᐸᒃᓯᓯᒪᓐᓂᖅᑎᓪᓗᒋᑦ ᐊᓯᕈᖅᑐ ᑕᒪᐃᓐᓂ ᐃᓚᒦᓂ ᑭᓪᓕᖏᓐᓂ ᐅᓄᕈᓐᓃᖅᑐᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᓄᖅᓯᒋᐊᖅᑐᑦ ᑖᒃᑯᐊ ᓇᓚᐅᑖᖅᓯᒪᔪᑦ ᐊᒻᒪᓗ ᓇᓐᓄᒃᑕᐅᓇᓱᒍᓐᓇᖅᑐᑦ ᑖᒃᑯᑎᒍᑦ ᐋᖅᑭᒃᑕᐅᖃᑦᑕᕐᒪᑕ.

ᑕᐃᑲᓂᓕᒫᖅ ᐃᓕᑦᑎᓇᓯᒃᑎᓪᓗᑕ, ᐊᔾᔨᒌᓕᕈᓐᓇᓚᐅᓐᖏᑦᑐᑦ ᐊᕙᑖᓂᑦ-ᐊᕐᕌᒍᒥ ᑕᐃᒪᓐᓇᐅᕙᖕᓂᖓ ᑕᕆᐅᖅ ᓯᑯᐊ. ᓇᓂᓯᓚᐅᖅᑐᒍᑦ ᐅᑯᐊ ᓇᓄᐃᑦ ᓴᖑᔪᑦ ᓯᒡᔭᒦᑕᖅᑐᑦ ᒪᓕᑦᑎᐊᖅᑐᑦ ᖃᓄᖅ ᓯᑯᓛᑦ ᐱᑕᖃᕐᒪᖔᑕ ᐱᓇᖕᓇᖓᓂ ᐅᕝᕙᓘᓐᓂ ᑲᓇᖕᓇᖓᓂ ᐃᒃᐱᐊᕐᔪᐊᑕ. ᑕᐃᒪᓗ ᑭᖑᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᑦ ᐃᑲᔪᖅᑐᑦ ᓇᓄᕐᓂᒃ ᑭᓪᓕᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. (ᓯᑑᓕᓐ ᐊᓯᖏᓪᓗ. 1977, ᑐᓚᐅᓱ ᐊᒻᒪᓗ ᓯᑑᓕᓐ 1990, ᐲᑳᒃ ᐊᓯᖏᓪᓗ. 2010, ᐊᑉᐹᑦ ᐊᒻᒪᓗ ᒥᑦᑎᐅ 2012), ᐅᕙᒍᑦ ᑐᑭᓯᓇᖅᓯᔪᑦ ᖃᐅᔨᓴᕈᑎᓂᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᑖᒃᑯᐊ ᑕᐃᒪᐃᑎᑦᑎᖕᒪᑕ ᓯᑯᐃᑦ ᖃᓄᐃᓕᖓᔭᕌᖓᒥᒃ ᓇᓄᐃᑦ ᓇᒧᖓᐅᕙᒋᓕᓲᖏᖅ, ᐃᓚᖏᓐᓂ ᐊᕐᕌᒍᓂ, ᓈᒻᒪᑦᑎᐊᕐᓗᑎᒃ ᑖᒃᑯᓄᖓ ᓴᖑᓂᖓ ᐊᓄᕐᓂᖏᓐᓄᒃ ᓇᓚᐅᑖᖅᓯᒪᔪᑦ ᐅᓄᖅᓯᕗᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᑦᑎᒃᓯᕗᑦ ᒪᓕᒃᖢᒍᓂ ᓄᒃᑕᕐᓂᖏᑦᑎᒍᑦ. ᐆᒃᑑᑎᒋᓪᓗᒍ , 2021-ᒥ, ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᓯᒪᔪᓂᒃ ᖃᖓᑕᓪᓗᑎᒃ ᑕᐃᑲᓂ ᓯᑯᓛᒃᑯᑦ ᑲᓇᖕᓇᖓᓃᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ ᑎᑎᕋᖅᑕᐅᓚᐅᖅᑐᑦ ᐊᓯᐊᓄᐊᕐᓂᖓᓂᑦ ᑕᐅᕙᖓᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐅᕙᖓ ᓂᒋᐊᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᑕᒪᓐᓇ ᓴᖑᓂᖓ ᐃᓗᐊᓃᓪᓗᐊᑕᖅᑐᖅ 200 ᑭᓚᒦᑕᔅ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑭᓪᓕᐊ ᐊᒻᒪᓗ ᐊᒥᓲᓐᖏᑦᑐᑦ ᓇᓄᐃᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᑕᐃᑲᖓᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐅᖓᓯᖕᓂᓴᐅᓂᖏᓐᓂᒃ ᑕᐅᕙᖓᒃᑲᓐᓂᖅ ᑲᓇᖕᓇᖓᓄᑦ.

ᐅᓄᓖᑦᑎᐊᖏᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᓇᓂᔭᐅᖃᑦᑕᖅᑐᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓵᓚᐅᖅᑐᖅ ᓄᓇ 2. ᑕᐃᑲᓂ 2022 ᑕᐃᒪᓗ ᓯᑯᒐᓖᓂᕐᒪᑦ ᐱᓇᖕᓇᖓᓂ ᐊᒻᒪᓗ ᑎᑎᕋᖅᓯᓚᐅᖅᑐᒍᑦ 36% ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᓯᑯᒐᓖᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᓚᐅᖅᑐᑦ 2022-ᒥ ᐅᑯᐊ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ. ᑲᒪᓇᓚᐅᖅᑐᖅ ᓇᓂᓯᓚᐅᓐᖏᓐᓇᑦᑕ ᐃᓐᓇᕐᓂᒃ ᐊᖑᑎᓂᒃ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐊᓯᖏᑦ ᑭᓯᐊᓂ ᐅᑭᐅᖃᑎᖏᑦ ᐊᒻᒪᓗ ᐊᕐᓇᐃᑦ ᐊᕙᑖᓂᑦ 2021 ᐊᒻᒪᓗ 2022 ᐱᑕᖃᓐᖏᒃᑲᓗᐊᕐᒪᑕ ᑭᒡᓕᖅᑐᐃᓯᒪᔪᑦ ᐊᒥᓲᓖᓂᒃ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᓐᓂᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᑕᐃᑲᓃᑦᑐᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ ᐊᒥᓱᑦ ᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᓯᒋᔭᒥᖕᓂ ᐊᑦᔨᒋᖏᑕᖓ ᖃᓂᓐᖓ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ ᐊᒻᒪᓗ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᑭᓯᐊᓂ ᖃᐅᔨᓇᓱᒃᓯᒪᓐᖏᑦᑐᒍᑦ ᖃᓄᑎᒋᖕᒪᖔᑕ ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᖑᑎᓄᑦ. ᑐᓚᐅᓯ ᐊᒻᒪᓗ ᓯᑑᓐ (1990) ᓇᓂᓯᓚᐅᖅᑐᑦ ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ ᓄᒃᑕᒃᑲᐅᓂᖅᓴᐅᔪᑦ ᐊᓯᖏᓐᓂᑦ ᓇᓄᕐᓂᑦ ᖃᓄᑐᐃᓐᓇᖅ ᐊᖑᑕᐅᓚᓗᐊᕈᑎ ᐃᓐᓇᕈᖅᐸᓪᓕᐊᓚᓗᐊᕈᑎᒃ (ᐊᓯᖏᑦ ᑕᐃᑲᓃᖏᓐᓇᖅᑐᓂᑦ ᐊᕐᓇᓂᑦ) ᐊᒻᒪᓗ ᑕᐃᒫᒃᑕᐅᖅ ᐅᓄᕐᓂᖅᓴᐅᔪᑦ ᐊᑯᓚᐃᑦᑐᒥᒃ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᑖᓄᐊᖅᑲᑦᑕᖅᑐᑦ 150 ᑭᓛᒥᑕᕐ ᐱᓂᖅᓴᐅᔪᑦ ᐊᓯᖏᓐᓂᑦ ᑕᒪᕐᒥᒃ ᐅᑭᐅᖃᑎᒋᓐᖏᑕᖏᑦ ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐃᓐᓇᕈᓕᖅᑐᓂᓪᓗ.

ᓇᓂᓴᑦᑎᓐᓂᑦ ᐅᓄᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ ᑖᒃᑯᐊᓗ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ ᐃᓚᖏᓐᓂ ᐊᕐᕌᒍᓂ ᐊᓯᒍᑦ ᐱᑕᖃᓐᖏᒃᑲᓗᐊᖅᑎᓪᓗᒋᑦ ᓇᓗᓇᐃᖅᑎᓐᓇᖅᑐᑦ ᐊᖏᓂᖅᓴᐅᓂᖓ ᐊᓯᐳᖅᑕᖅᑕᖅᓂᖏᑦᑕ ᓄᒃᑕᖅᑐᖅ/ᑎᑭᑦᑐᓪᓗ ᑕᐃᒪᓐᓇᐅᓂᖅᓴᐅᓕᖅᑐᑦ ᑭᖑᓂᑦᑎᓐᓇᓂ ᖁᓕᓂᑦ ᐊᖅᑯᓂ (ᓚᓐ ᐊᓯᖏᓪᓗ. 1997). ᑕᐃᒪᓐᓇ ᐱᔾᔪᑎᖃᕐᓂᖓᓄᑦ ᖃᐅᔨᒪᓐᓇᖏᑦᑐᖅ ᑭᓯᐊᓂ ᐊᑕᔪᓐᓇᖅᑐᑦ ᐊᖏᔫᓂᖓᓂ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᖏᓐᓇᖅᑐᑦ, ᐊᓯᑭᖅᑕᓖᑦ ᖃᓄᓇᑑᓂᖏᑦ ᑕᕆᐅᖅ ᓯᑯᐊ, ᐊᓯᖑᖅᑎᑕᐅᒃᐸᑦ ᐃᓂᒋᔭᖏᑦ ᐊᑐᖅᐸᒃᑕᖏᑦ, ᐊᔾᔨᒋᔭᖕᒋᑕᖏᑦ, ᑕᒫᓂᐸᓪᓕᖅᑐᑦ ᐊᒻᒪᓗ ᓇᒍᓕᓯᒪᖏᑦ ᓴᖑᔪᑦ ᓇᓄᐃᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓯᑯᖃᓐᖏᓐᓂᖓᓄᑦ (ᑕᐅᓐ ᐊᓯᖏᓪᓗ. 2010; ᒥᒃᑳᓐ ᐊᓯᖏᓪᓗ. 2015, ᓄᐊᑦᑐᓚᑉ ᐊᓯᖏᓪᓗ. 2022). ᐃᒪᖄ, ᓱᕋᑦᑎᓂᖓ ᖃᐅᔨᒪᓐᓇᖏᑦᑐᒥᒃ ᓯᓚᐅᑉ ᖃᓄᐃᓐᓂᖓᒍᑦ ᑕᐃᒪᐃᑎᑦᑎᔪᑦ ᓇᓄᐃᑦ ᓇᒧᖓᖅᑕᖅᑐᑦ. ᐆᒃᑑᑎᒋᓪᓗᒍ, ᐊᕐᕌᒍᓂ ᑭᖑᐊᖅᓯᒪᓪᓗᑎᒃ ᓯᑯᐃᔭᖃᑦᑕᖅᑐᑦ ᐊᒃᑐᐃᔪᓐᓇᖅᑐᑦ ᓇᒧᖓᔪᒪᓂᖓᓂᒃ ᓇᓄᐃᑦ ᑕᐃᒪᓗ ᖃᖓᓗᐊᖅ ᑎᑭᑲᑦᑕᖅᐸᑕ ᓯᒡᔭᓄᐊᖅᐸᑕ ᐊᒻᒪᓗ ᑎᑭᓚᐅᖅᑐᑎᒃ ᐅᐸᒍᒪᔭᒥᖕᓄᑦ ᐊᐅᔭᕕᒋᔭᒥᖕᓄᑦ ᓄᓇᒥ (ᓯᑑᓐ ᐊᓯᖏᓪᓗ.

2004). ᐆᒃᑑᑎᒋᓪᓗᒍ, ᐊᕐᕌᒍᑦ ᓯᑯᐃᕐᓇᓵᕐᓂᕌᖓᑦ ᐊᒃᑐᐃᔪᓐᓇᖅᑰᖅᑐᑦ ᓇᓄᕐᓂᒃ ᓇᒧᖕᖒᓯᒪᓂᖏᓐᓂᒃ ᑕᐃᒪᓗ ᒪᓐᓇᑖᖅ ᓯᒡᔭᒧᐊᖅᑐᑦ ᐊᒻᒪᓗ ᓱᓕ ᑎᑭᓐᖏᑦᑐᑦ ᐊᐅᔭᕐᕕᒋᔦᒪᔭᕐᒥᖕᓄᑦ ᓄᓇᒧᑦ (ᓯᑑᓕᖕ ᐊᓯᖏᓪᓗ. 2004). ᑐᓚᐅᓯ ᐊᓯᖏᓪᓗ. (2004) ᐃᓱᒪᓕᖅᓯᓯᒪᔪᖅ ᐅᑯᐊ ᑲᑎᖕᖓᐅᓕᖅᑐᑦ ᐃᒻᒥᖕᓄᑦ ᐃᓚᒌᑦ ᐅᓄᖅᓯᔪᓐᓇᖅᑐᑦ ᐊᓯᕈᖅᑐᓄᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᖅ ᓯᑯᐊ ᓇᒧᖕᖒᓯᒪᓂᖓ. ᓇᓂᔭᕗᑦ ᐃᑲᔪᖅᑐᑦ ᑕᒪᑐᒥᖓ ᑕᐃᒪᓐᓇᐅᓕᐊᕋᓱᒋᓚᐅᖅᑕᑦᑎᓐᓂᒃ. ᓱᑯᐃᔭᕌᖓᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓯᑯᐃᖅᖄᓕᓪᓗᓂ (ᓯᑑᓐ ᐊᒻᒪᓗ ᓚᐃᑐᓕ 2016), ᐊᒻᒪᓗ ᓇᓄᐃᑦ ᓯᒡᔪᒃᑯᑦ ᑎᑭᖃᑦᑕᖅᑐᑦ ᓯᑯᖃᑦᑎᐊᓐᖏᑎᓪᓗᒍ ᓕᖕ ᐊᓯᖏᓪᓗ. 1999). ᑕᐃᒪᓐᓇᐅᓚᐅᕐᒪᑦ, ᓇᓄᐃᑦ ᓵᑎᓯᒪᓕᕈᓐᓇᖅᑐᑦ ᐊᓯᒥᐅᑕᒥᖕᓂᒃ ᑎᑭᐅᕋᖅᑐᑦ ᐅᑯᐊᓗ ᖁᐃᓂᖏᓐᓂᖅᓴᐃᑦ ᑭᓯᐊᓂ ᐊᔾᔨᐸᓗᖕᒋᑦ ᑖᒃᑯᐊ ᓄᓇᐃᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᔾᔨᐸᓗᖕᒋᑦ ᓯᖃᓐᖏᓂᖕᖓᓄᑦ ᖁᐃᒃᑎᑦᑐᓐᓇᐃᓕᔪᑦ ᑭᓯᐊᓂ ᓄᓇᒧᐊᖃᑦᑕᓕᖅᑐᑦ ᓯᒡᔪᒃᑯᑦ ᐊᓯᐊᓄᑦ.

ᑕᐃᑲᓂ 2021 ᖃᖕᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ 27% ᓄᖑᐹᓪᓕᖅᓯᒪᓚᐅᖅᑐᑦ ᑕᐃᑲᖓᑦ 2016 ᓇᓚᐅᑖᖅᓯᒪᔪᓂᑦ ᐅᑯᐊᓗ ᑕᐃᒪᓐᓇᐅᔪᑦ ᐅᓄᓐᖏᓂᖅᓴᐅᖕᒪᑕ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᐊᒻᒪᓗ ᐃᓐᓇᕈᓕᖅᑐᑦ (ᐊᑦᑭᓐᓴᓐ ᐊᓯᖏᓪᓗ. 2022). ᓇᓂᓯᓚᐅᖅᑐᒍᑦ ᑕᑯᑎᑕᐅᓪᓗᑕ ᕿᓚᒥ ᐊᐅᓪᓚᖅᓯᒪᓚᐅᒃᑐᑦ ᑕᐃᑲᖓ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ 2021-ᒥ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᓯᕈᕐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ ᐅᑎᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒧᑦ ᑕᐃᑲᓂ 2022. ᓅᓴᖅᓗᒋᑦ ᕿᓚᒥ ᑎᑭᑉᐸᒃᑐᑦ/ᐊᐅᓪᓚᖅᐸᒃᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᑕᐃᑲᓂ 2022, ᐃᓕᓐᓇᓱᒃᑕᕗᑦ ᐃᓱᒪᓕᖅᓯᓯᒪᔪᑦ ᑖᒃᑯᐊ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐅᓄᖅᓯᕆᐊᕋᔾᔪᐊᖅᑐᑦ 29% ᓄᒃᑕᖅᑕᖏᓐᓂᑦ ᑖᒃᑯᐊᑐᐊᖅ ᑕᐃᒪᓗ ᐊᔾᔨᑎᐊᕆᔾᔪᖓ ᐊᓯᕈᖅᑐᓄᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (27%). ᐊᒻᒪᓗᒃᑲᓂᖅ, ᑕᐃᑲᓂ ᑲᑎᓪᓗᒋᑦ ᓇᓚᐅᑖᓯᒪᔪᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᑲᓂ 2016 ᖃᓅᑎᒋᖕᒪᖖᑕ ᑖᒃᑯᓇᖓᑦ 2021-ᒧᑦ, ᐊᓯᕈᖅᑐᖃᖕᒋᑦᑐᖅ ᐅᓄᕐᓂᖏᓐᓂᒃ (ᐊᑦᑭᓐᓴᓐ ᐊᓯᖏᓪᓗ. 2022). ᑕᐃᒪᓗ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓪᓗᑎᒃ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᑖᒃᑯᐊᑐᐊᓪᓚᐅᑕᐅᓚᐅᖅᑐᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, ᐃᓕᓐᓇᓱᒃᑕᑦᑎᓐᓂ ᓴᓇᓯᒪᓂᖓ ᓇᓂᓱᒃᑲᓂᕈᓐᓇᕋᔭᓚᐅᖅᑐᑦ ᐃᓐᓇᕐᓂᒃ ᐊᖑᑎᓂᒃ ᐊᐅᓪᓚᑎᓗᒋᑦ ᓯᓚᔾᔭᒋᑦ ᓂᒋᐊᑕ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᐃᓕᓐᓇᓱᒃᑕᕗᑦ ᖃᐅᔨᓚᐅᕐᖏᑦᑐᑦ ᐅᓄᕐᓂᖅᓴᐅᖕᒪᖖᑕ ᓇᔪᖅᐸᒃᑕᒥᖕᓂᕐᖏᑦᑐᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ ᖃᓅᑎᒋᓂᖏᑦ ᐊᓯᖏᓐᓂᑦ ᐅᑭᐅᖃᑎᒥᖕᓂᒃ ᐊᒻᒪᓗ ᓇᓂᓯᓚᐅᖅᑐᒍᑦ ᑲᒪᓇᖅᑐᒥᒃ ᐅᓄᖅᑐᐊᓘᖕᒪᑕ ᓇᔪᖅᐸᒃᑕᒥᖕᓂᕐᖏᑦᑐᑦ ᐃᓐᓇᕈᓕᖅᑐᓄᑦ

ᐊᕐᓇᓄᑦ. ᑕᑯᓪᓗᒍ ᑖᓐᓇ ᓄᓇ 2 ᑖᒃᑯᐊᑦᑕᐃᓐᓇᓪᓗᐊᑕᐅᖓᑕ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ, ᑕᐃᒪᓗ ᐃᓐᓇᖅᑕᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐊᑯᑎᓂᒃ ᑕᐃᒪᖓᓂᑦ 2017 ᐊᒻᒪᓗ ᑕᐃᒪᐃᒻᒪᑦ ᓇᓂᓱᒃᑳᓂᕐᓂᐊᖅᑐᒍᑦ ᑖᒃᑯᐊ ᐅᑭᐅᓂᑦ ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐊᕐᓇᓗ. ᑕᐃᒪᓐᓇᐅᖃᓗᐊᕐᒪᑦ, 16% ᑖᒃᑯᐊᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᖃᐅᔨᓴᕈᑎᑖᕋᓱᒃᑲᓗᐊᖅᑎᓪᓗᑕ ᓯᒃᑐᐊᖓᓂᑦ, ᐅᓄᕐᓂᖅᓴᐅᖏᓐᓇᔭᓚᐅᖅᑐᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ. ᐊᔾᔨᒌᓐᖏᓂᖏᑦ ᓄᓇᖃᑎᒌᓐᖏᑦᑐᑦ ᑎᑎᕋᖅᑕᐅᓚᐅᖅᑐᖅ ᐊᕙᑖᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᐅᓄᕈᓐᓃᖅᓯᒪᔪᑦ ᑎᑎᕋᖅᑕᐅᓚᐅᖅᑐᑦ ᑕᐃᑲᓂᓪᓗᐊᑕᖅ ᓄᓇ 1 ᐃᓱᒪᓕᖅᓯᓯᒪᔪᖅ ᐅᓄᕈᓐᓃᓕᕐᓂᖏᓐᓂᒃ ᓇᓄᐃᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐊᒻᒪᓗ ᐅᓄᖅᓯᓪᓗᑎᒃ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑲᑎᖓᓪᓗᑎᒃ ᑎᑭᓚᐅᑲᒃᓯᒪᔪᑦ/ᐊᐅᓪᓚᖅᐸᓪᓕᐊᔪᑦ ᑕᐃᒫᒃ ᐊᒥᓲᓐᖑᓗᐊᕐᑰᔾᔮᑦ ᓇᓚᐅᑖᖅᓯᒪᔪᑦ ᑖᒃᑯᓄᖓ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᐊᒻᒪᓗ ᐅᒥᓱᕈᖅᓯᒪᓕᕐᒪᑕ ᓇᓚᐅᑖᖅᓯᒪᔪᓂᑦ ᑖᒃᑯᐊ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᐊᑦᑭᓐᓴᓐ ᐊᓯᖏᓪᓗ. 2022, ᓄᐊᑦᑐᓚᑉ ᐊᓯᖏᓪᓗ. 2022).

ᓇᒧᖕᒌᓯᒪᕙᓕᕐᓂᖏᑦ ᓇᔪᖅᐸᒃᑕᖏᓐᓂᐅᓐᖏᑦᑐᖅ ᑕᐃᒪᓐᓇᐅᔪᖅ ᓇᒥᑐᐃᓐᓇᖅ ᐃᓚᒌᑦ ᓇᓄᐃᑦ ᑭᒡᓕᐊᓂ ᑕᐃᒪᓗ ᑕᒪᕐᒥᒃ ᓇᓄᐃᑦ ᓈᓴᖅᑕᐅᓯᒪᕚᑦ ᐊᑕᐅᓯᐊᖅᖢᑎᒃ ᖃᖓᑕᔫᒃᑯᑦ ᐊᑕᐅᓯᖅ ᐃᓚᖏᑦ ᑕᒪᐃᓐᓂᖒᖅᓯᒪᕐᑰᓐᖏᑦᑐᑦ. ᑕᒪᕐᒥᒃ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᖏᑦ ᓇᓐᓄᒐᓱᒃᑕᐅᔫᑦ, ᐊᒻᒪᓗ ᓂᕿᒃᓴᕆᔭᐅᓪᓗᑎᒃ ᖃᐅᔨᒪᓇᓕᔫᖅ ᐅᓄᕐᓂᖏᑦᑎᒍᑦ ᒪᓕᒃᓗᒋᑦ ᖃᖓᑕᔫᒃᑯᑦ ᓈᓴᖅᑕᐅᔪᓂᑦ ᖃᖓᑕᓪᓗᑎᒃ. ᐃᓕᓐᓇᓱᒃᑕᕗᑦ ᓴᕿᑎᑦᑎᔪᖅ ᐅᓄᖅᓯᒋᐊᖅᑎᑦᑎᓯᒪᔪᓐᓇᖅᑐᖅ ᐊᒻᒪᓗ ᐅᓄᓐᖏᒡᓕᒋᐊᖅᑎᑦᑎᓯᒪᔪᓐᓇᖅᑐᓗ ᐊᖑᓇᓱᒃᑕᐅᔪᓂ ᖃᓅᑎᒋᓂᖏᓐᓂ ᑕᒫᓐᖓᑦᑐᓄᑦ ᕿᓇᒥ ᐅᐸᒃᓯᒪᓚᐅᖃᒃᑕᖏᓐᓂᑦ ᓇᓄᐃᑦ. ᐊᒥᓱᑦ ᓈᓴᖅᓯᒪᔪᑦ ᓇᓚᐅᑖᖅᓯᒪᔪᑦ ᐱᓱᒪᓕᖅᑎᑦᑎᔪᓐᓇᖅᑐᑦ ᐊᖑᓇᓱᒃᑕᐅᓗᐊᖅᑎᑦᑎᓗᑎᒃ ᐅᕝᕙᓘᓐᓃᑦ ᐊᖑᓇᓱᒃᑕᐅᓗᐊᖅᑎᑦᑎᓐᖏᓪᓗᑎᒃ ᐊᐅᓚᑦᑎᔾᔪᑎᒃᓴᓄᑦ ᑐᑭᓕᐅᕈᑎᒋᓯᒪᔭᑎᒃ ᐊᑐᖅᑕᐅᓂᐊᕐᒪᑕ. ᑕᐃᒪᖓᓂᑦ ᑕᒪᕐᒥᐸᓗᒃ ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᓂᕕᐊᑖᓕᖅᓯᒪᓐᖏᑦᑐᑦ, ᓇᓕᖃᓅᓕᖅᑎᑦᑎᓂᖅ ᐊᐅᖏᑦᑎᒍᑦ ᑭᓇᐅᓂᖏᓐᓂᒃ ᑖᒃᑯᓇᖓᑦ ᓂᕿᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕕᐅᓯᒪᔪᓂᑦ ᓂᕕᖓᑖᖃᓐᖏᑦᑐᑦ ᓇᓐᓄᒃᑕᖑᔪᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᑦ ᑐᓴᐅᒪᔾᔪᑎᒃᓴᑐᐃᓐᓇᐅᓂᐊᖅᑐᑦ ᖃᐅᔨᓇᓱᑦᑎᐊᕐᓗᑦ ᐊᑦᑕᕐᓇᖅᑑᒥᒻᒪᖔᑕ ᓇᓐᓄᒃᑕᐅᓇᓱᒋᐊᒃᓴᖏᓐᓂᒃ ᐊᔾᔨᒌᓐᖏᑕᖓ ᖃᓂᒋᔭᖏᓐᓂᑦ ᓄᓇᓂᑦ. ᑕᐃᒪᓐᓇᐅᖃᓗᐊᖅᑎᓪᓗᒍ, ᐅᓇ ᐃᓕᑕᐅᓇᓱᒃᑐᖅ ᒪᓕᑦᑎᐊᕋᔭᖅᑑᖃᓗᐊᖅ

ᑐᓴᒐᒃᓴᓂᒃ ᑐᓂᔭᐅᔪᓂᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᕕᓂᕐᓂᑦ ᑐᑭᓕᐅᕈᑎᒋᓗᒍ ᖃᓄᐃᓕᖓᓂᖓ ᐊᒻᒪᓗ ᐊᓯᕈᖅᑕᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᑕᒪᐃᓐᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ.

ᐃᓕᓴᕆᔭᐅᔪᑦ

ᖃᓄᐃᓕᐅᕆᐊᖃᖅᑐᓄᑦ, ᑮᓇᐅᔭᓂᒃ, ᐊᒻᒪᓗ ᖃᓄᑐᐃᓐᓇᖅ ᐃᑲᔪᖅᑐᑦ, ᖁᔭᓐᓇᒦᒃ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐅᑭᐅᖅᑕᖅᑐᒥ ᐃᓕᓐᓂᐊᕈᑎᖏᑦ, ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑲᓇᑕ, ᑲᔅᑲ ᒍᔅ ᓯᓂᒃᑕᕐᕕᒃ, ᐃᔅᑎᐅ ᐃᓚᒦᑦ ᑐᓐᖓᕕᖓᑦ, ᒫᓂᑑᐸ ᓄᓇᒥᐅᑕᓕᕆᔨᒃᑯᑦ, ᓄᓇᒥ ᖃᐅᔨᓴᖅᑎᑦ ᐊᒻᒪᓗ ᐃᑯᒪᓕᕆᓂᕐᒥ ᖃᐅᔨᓴᖅᑐᑦ ᑲᑎᒪᔨᖏᑦ ᑲᓇᑕᒥ, ᐋᓐᑎᐅᕆᐅ ᒥᓂᔅᑕ ᓄᓇᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᐊᒻᒪᓗ ᓇᐹᖅᑐᓕᕆᓂᕐᒧᑦ, ᒥᕐᖑᐃᖅᓯᕐᕕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕ, ᓇᓄᐃᑦ ᓯᓚᕐᔪᐊᖅ, ᑲᓇᖕᓇᖓᓂ ᖁᓕᒥᒎᓕᑦ, ᑕᒃᓚ ᑐᓐᖓᕕᖓ, ᐃᓕᓐᓂᐊᕐᕕᒃᔪᐊᖅ ᐊᐃᐳᑕ, ᐆᒪᔪᓕᕆᓂᖅ ᐱᔨᑦᑎᕋᖅᑎᑦ ᑲᓇᑕ, ᒍᐃᔅᑕᓐ ᑐᓐᖓᕕᖓ, ᐆᒪᔪᓕᕆᔪᑦ ᑐᓴᒐᒃᓴᓕᕆᔨᑦ, ᑎᒥᖓ, ᐊᒻᒪᓗ ᓯᓚᕐᔪᐊᒥ ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᑲᓇᑕ (ᐅᑭᐅᖅᑕᖅᑐᒥ ᐆᒪᔪᖏᓐᓂᒃ ᓴᐳᓐᓂᐊᕈᑎᑦ ᑮᓇᐅᔭᒃᓴᖅ). ᖁᔭᓐᓇᒦᒃ ᒍᓕᒃ ᑎᒫᓐ, ᓚᐃᔪ ᒍᐊᑕᓐ, ᐊᒻᒪᓗ ᑕᐃᓗ ᐅᓖᔅ ᐃᑲᔪᓚᐅᕐᒪᑕ ᓄᓇᒥ. ᑯᔭᓐᓇᒦᒃ ᖃᖓᑕᓲᖅᑎᑦ ᑕᒻ Hᐅᑦᐱ (OMNRF), ᐊᒻᒪᓗ ᔭᐃᒥ ᐳᓪᔅ ᐊᒻᒪᓗ ᔭᔅᑎᓐ ᓯᓂᐅᒃ (ᐱᓇᖕᓇᖓᓂ ᖁᓕᒥᒎᖅᑎᑦ). ᖁᔭᓐᓇᒦᒃ ᐃᕙᓐ ᐅᓕᑦᓯᑦᓯᓐ ᐊᒻᒪᓗ ᓃᓐ ᐹᓐᔅ ᕿᒥᕐᕈᐊᓚᐅᕐᒪᑕ ᑭᖑᓂᑦᑎᓐᓂ ᐅᓂᒃᑳᓕᐅᖅᓯᒪᔭᑦᑎᓐᓂᒃ. ᐃᓕᓴᕆᔮᑦ ᐊᒻᒪᓗ ᖁᔭᓐᓇᒦᒃ ᐸᒃᔅ ᓕᒃ ᑯᓕᐄ ᐊᓪᓚᐃᑦ, ᒍᐊᔅHᐅ ᑯᓕᐄ, ᒍᐃᓅᔅ ᐊᓪᓚᐃᑦ ᐊᒻᒪᓗ ᔪᐊᒃ ᕚᒃᑐᓕ ᐊᓪᓚᐃᑦ.

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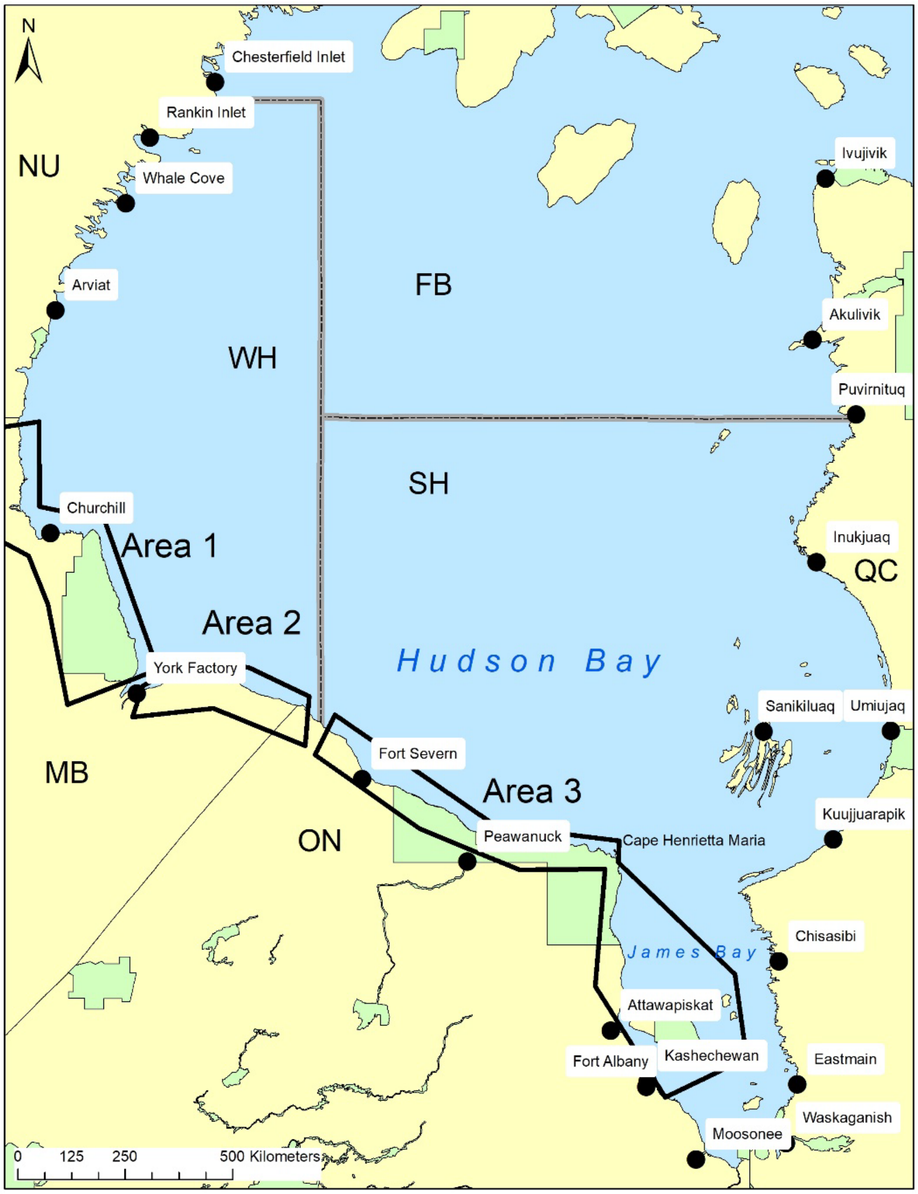
ᓇᓗᓇᐃᖅᓯᔪᖅ 1. ᖃᑦᓯᐅᓂᖏᑦ ᓂᕿᖏᑦᑎᒍᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᓇᓄᐃ >1 ᐅᑭᐅᖃᖅᑐᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᐊᔾᔨᒋᓚᐅᓐᖏᑕᖏᑦ ᓂᕕᖓᑖᓕᖅᑕᐅᓂᑯᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᔾᔨᒋᓚᐅᖅᑕᖏᑦ ᓂᕕᖓᑖᓕᖅᑕᐅᓂᑯᓂᑦ ᓇᓄᐃᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐅᕝᕙᓘᓐᓃᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᑭᖑᓂᑦᑎᓐᓂ ᓂᕕᖓᑖᓕᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᐃᓱᒪᖏᔭᐅᕗᑦ ᐱᒋᔭᐅᔪᑦ ᐃᓚᒌᖕᓂᑦ ᐅᑯᐊᓗ ᓯᕗᓪᓕᖅᐹᒥᒃ ᑕᑯᔭᐅᓚᐅᖅᑐᑦ. ᓄᓇ 3 ᖃᐅᔨᓴᕈᑎᑖᕐᕕᖓ ᓱᓕ ᐃᓗᐊᓃᖀᒃᔪᒃᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ/ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑭᓪᓕᐊ ᑕᐃᑲᓂ 2018/2019 ᐊᒻᒪᓗ ᑕᒪᕐᒥᐸᓗᒃ ᐋᓐᑎᐅᕆᐅ ᓯᒡᔭᖓ ᑕᓯᐅᔭᕐᔪᐊᒥ 2021-ᒥ ᐊᒻᒪᓗ ᐱᔭᓈᓚᐅᖅᑐᑦ ᑕᒪᒃᑭᖅᓱᒍ 2022-ᒥ.

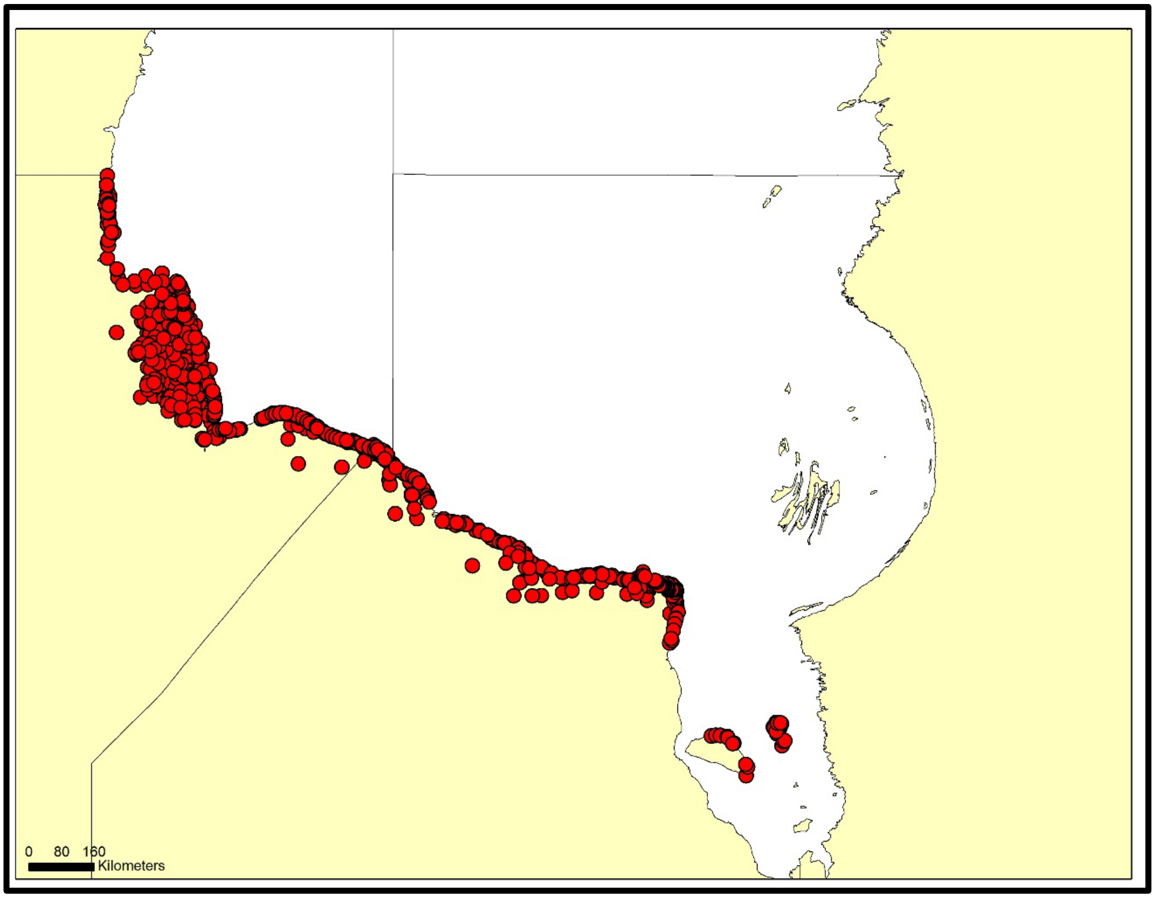
ᖃᐅᔨᓴᕈᑎᑖᕕᒃ ᓇᓃᓐᓂᖓ	ᐊᕐᕌᒍᒥ	ᖃᑦᓯᑦ	ᐊᔾᔨᒋᓚᐅᓐᖏᑕᖏᑦ ᓂᕕᖓᑖᓕᖅᑕᐅᓂᑯᓂᑦ	ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ	ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓄᐃᑦ
ᓄᓇ 2	2017†	60	34 (57%)	17 (28%)	9 (15%)
	2018	140	88 (63%)	32 (23%)	20 (14%)
	2019	160	113 (71%)	30 (19%)	17 (11%)
	2021*	140	44 (31%)	87 (62%)	9 (6%)
	2022*	146	33 (23%)	98 (67%)	15 (10%)
ᓄᓇ 3					
	2017	n/a	n/a	n/a	n/a
	2018	8	6 (75%)	1 (12.5%)	1 (12.5%)
	2019	12	7 (58%)	1 (8%)	4 (34%)
	2021*	183	119 (65%)	45 (25%)	19 (10%)
	2022*	259	182 (70%)	18 (7%)	59 (23%)

†ᖃᐅᔨᓴᕈᑎᑖᖅᑐᖃᓚᐅᓐᖏᑦᑐᖅ ᐋᓐᓂᐅᑎᓇᔾᔭᖓᓂ ᐃᓚᖓᓂ ᓄᓇ 2 ᑕᐃᑲᓂ 2017.

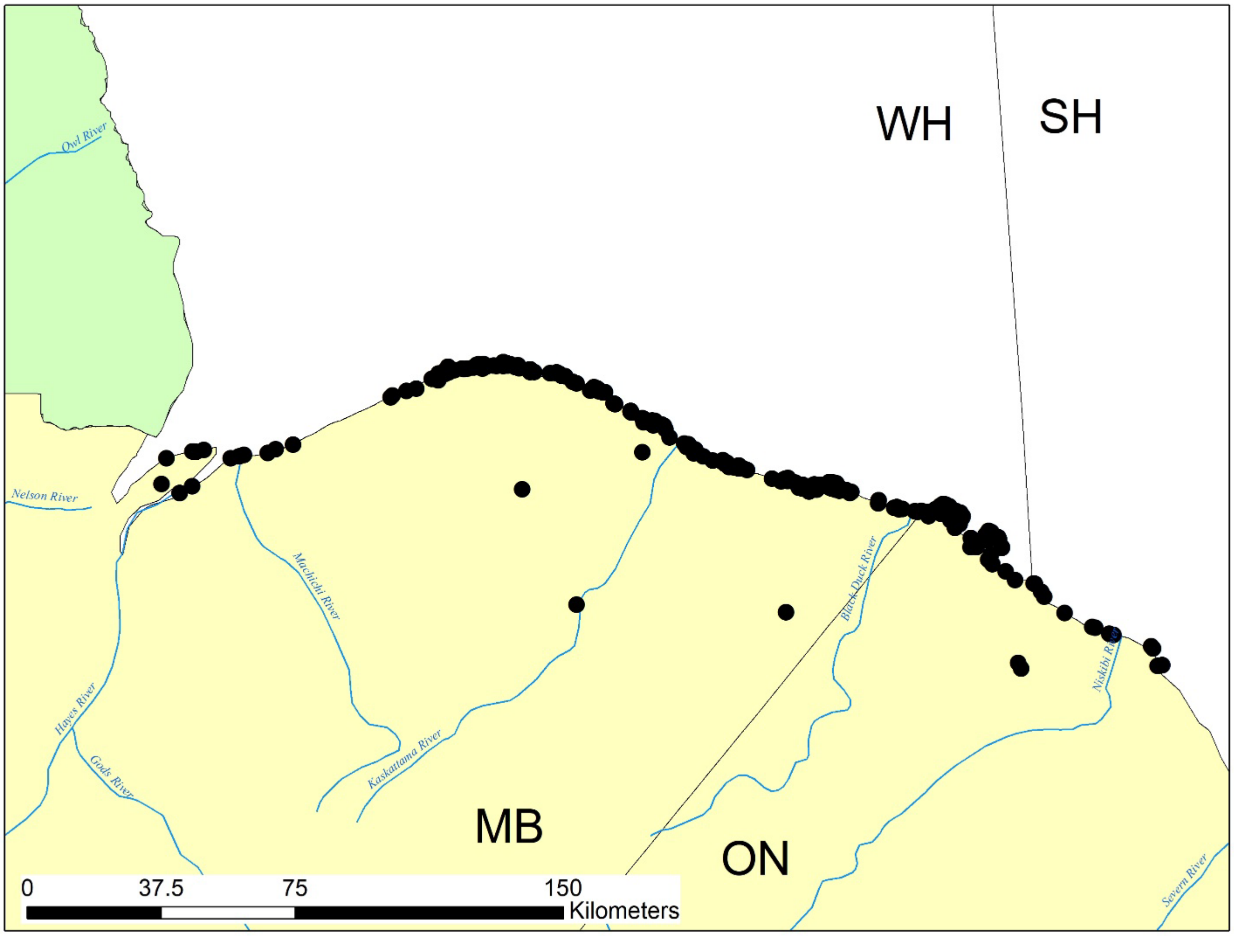
*2021 ᐊᒻᒪᓗ 2022 ᑕᒪᕐᒥᒃ ᑭᖑᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᐊᑐᖅᑕᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓇᓱᒃᓱᒋᑦ ᐊᐅᖏᑦᑎᒍᑦ ᐊᔾᔨᒋᔭᐅᖕᒪᖕᒌᑕ ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᓇᓄᐃᑦ ᑖᒃᑯᐊᑐᐊᖅ ᑭᖑᓂᑦᑎᓐᓂ ᓂᕿᖏᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓂᑯᑦ 2017-ᒥᑦ 2019-ᒧᑦ ᐊᔾᔨᒋᔭᑐᐊᖏᑦ ᓂᕕᖓᑖᖅᑐᓄᑦ ᓇᓄᕐᓄᑦ ᐅᖃᐅᓯᐅᓯᒪᔪᓂ.



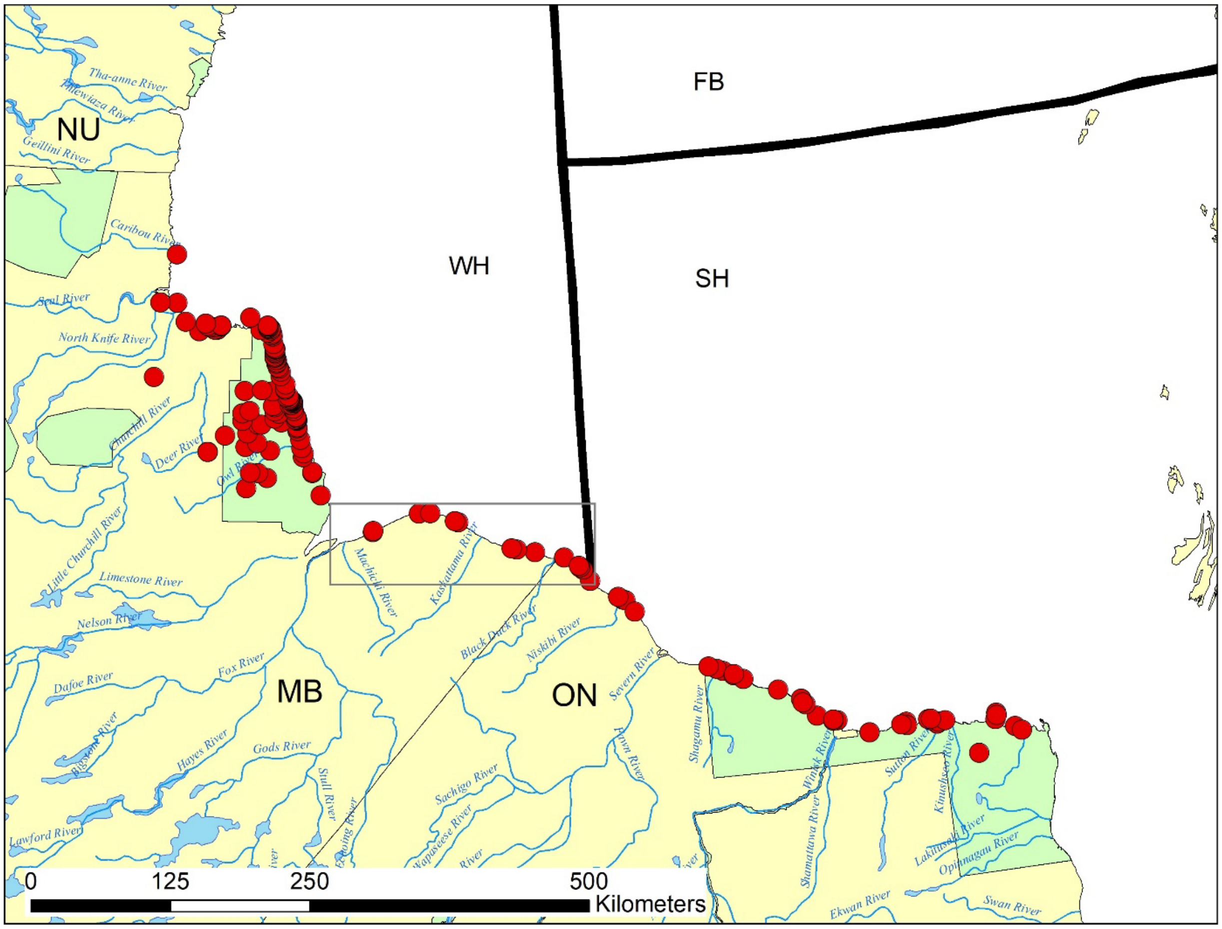
ᓴᕿᔮᖅᑐᖅ 1. ᑕᓯᐅᔭᕐᔪᐊᖅ ᓴᕿᔮᖅᑐᑦ ᐊᐅᓚᑦᑎᕕᐅᔪᑦ ᑭᒡᓕᖏᑦ (ᐊᖕᒪᓗᖅᑐᑦ) ᐊᕙᑖᓂᑦ ᐊᑭᓐᓇᖅᐸᓯᐊ ᑕᓯᐅᔭᕐᔪᐊᖅ, ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ, ᐊᒻᒪᓗ ᓯᑯᓯᓛᖅ ᐃᒪᖓ (FB) ᓄᓇᖏᑦ ᓇᓄᐃᑦ ᐃᓚᖏᓪᓗ ᐊᒻᒪᓗ ᓄᓇᖓᓂᒃ ᖃᐅᔨᓴᕈᑎᑖᑦ 1, 2 ᐊᒻᒪᓗ 3.



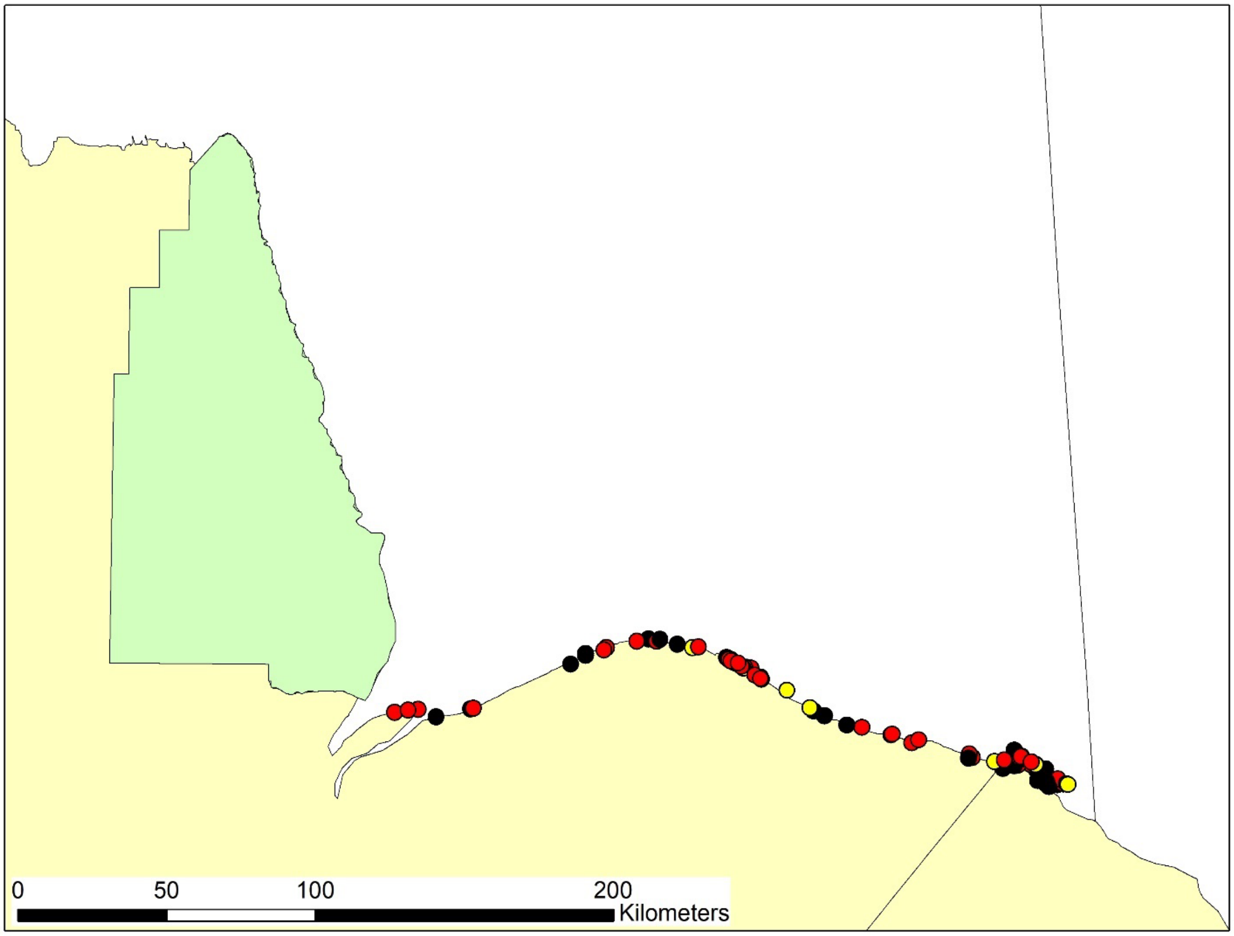
ᓴᕿᔮᖅᑐᖅ 2. ᐅᖕᒐᓯᒌᒃᑑᑕᐅᓂᖏᑦ ᓇᒧᖕᒑᓯᒪᔪᑦ ᓇᓄᐃᑦ ᑕᖕᒋᖕᒋᓐᓂᒃ ᖃᐅᔨᓴᕐᑎᑖᕐᕕᒋᓯᒪᔭᕗᑦ ᓂᕿᖕᒋᓐᓂᑦ ᖃᐅᔨᓴᕐᑎᑖᑦᑎᓐᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓇᓐᓄᒃᑕᐅᔪᓂᑦ ᐊᕙᑖᓂᑦ 2017 ᐊᒻᒪᓗ 2022.



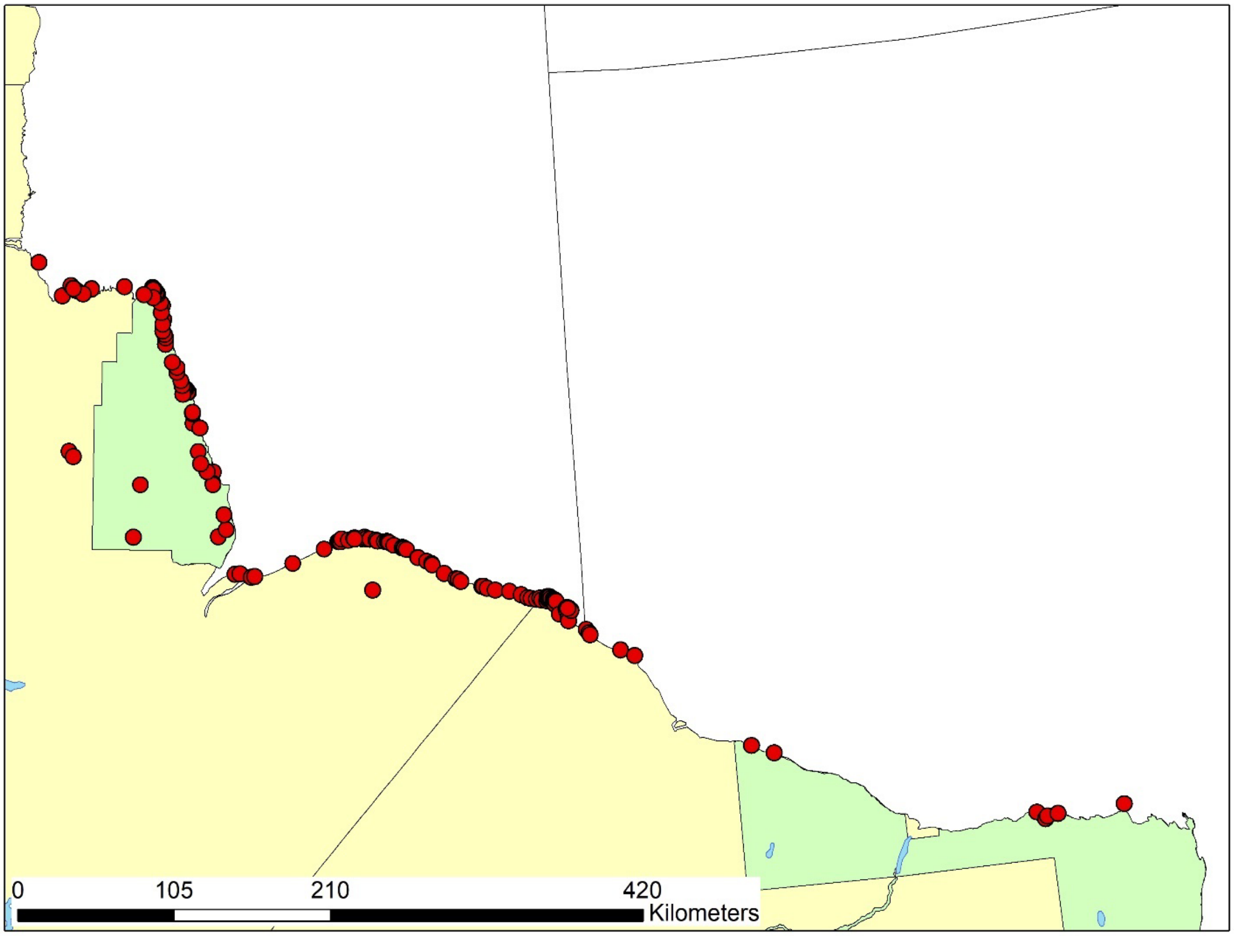
ᓴᕿᔮᖅᑐᖅ 3. ᓇᓅᓐᓂᖏᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᕈᑖᕐᕕᐅᔪᑦ, 2017-ᒥᑦ 2019-ᒧᑦ ᑖᒃᑯᓇᓂ ᓄᓇ 2 ᐊᒻᒪᓗ ᓄᓇ 3.



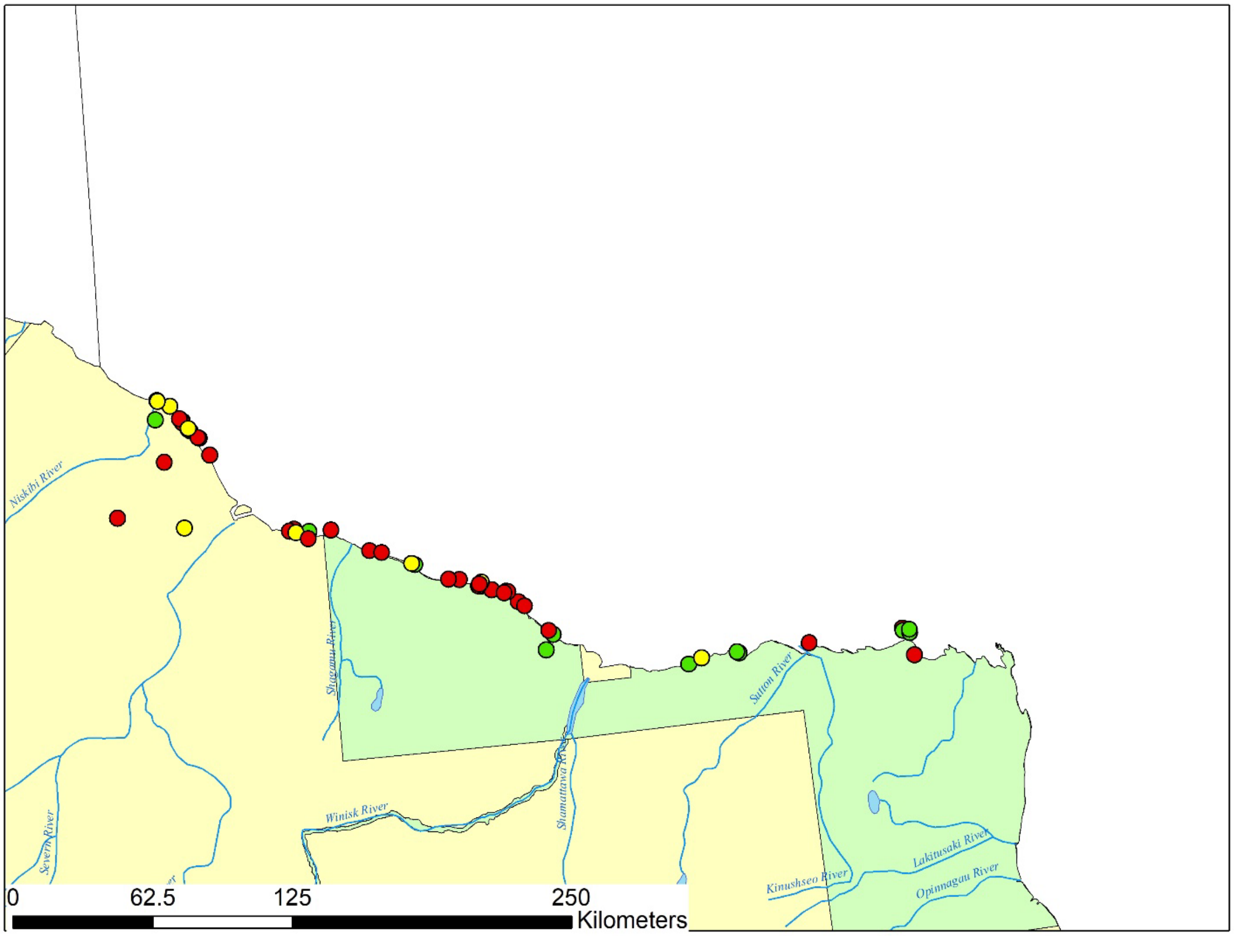
ᓴᕿᔮᖅᑐᖅ 4. ᑭᖑᓂᑦᑎᓐᓂ ᓇᓃᓚᐅᕐᓂᖏᑦ 100 ᓇᓄᐃᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᑕᐃᑲᓂ ᓄᓇ 2 (ᓯᐊᕐᓇᖅ ᑭᑉᐹᓗᑐᖅ) ᑕᐃᑲᙵᑦ 2017 ᑎᑭᓪᓗᒍ 2019 (ᓴᕿᔮᖅᑐᖅ 3) ᐅᓇ ᐊᔾᔨᙳᙱᑦ ᐊᑐᓂ ᓂᕕᙵᑖᖃᖅᑐᑦ ᓯᕗᓂᐊᓂ 2017 ᐊᒻᒪᓗ ᓇᓄᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᑕᐃᑲᙵᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᐅᓐᓄᑦ ᓇᓄᓇᐃᑯᑕᖅᕐᕕᐅᔪᒥ. ᓇᓐᓄᒃᑐᖃᓚᐅᙱᑦᑐᖅ ᐊᒻᒪᓗ ᓂᕕᙵᑖᓕᖅᓯᔪᖃᓚᐅᙱᑦᑐᖅ ᐋᓐᑎᐅᓇᐅ ᓯᓪᔭᓕᒫᙵᓂᑦ ᑕᐃᒪᙵᓂᑦ 2009 ᐅᕝᕙᓘᓐᓃᑦ ᑕᐃᑲᓂ ᓄᓇ 2 ᑕᐃᒪᙵᓂᑦ 2005.



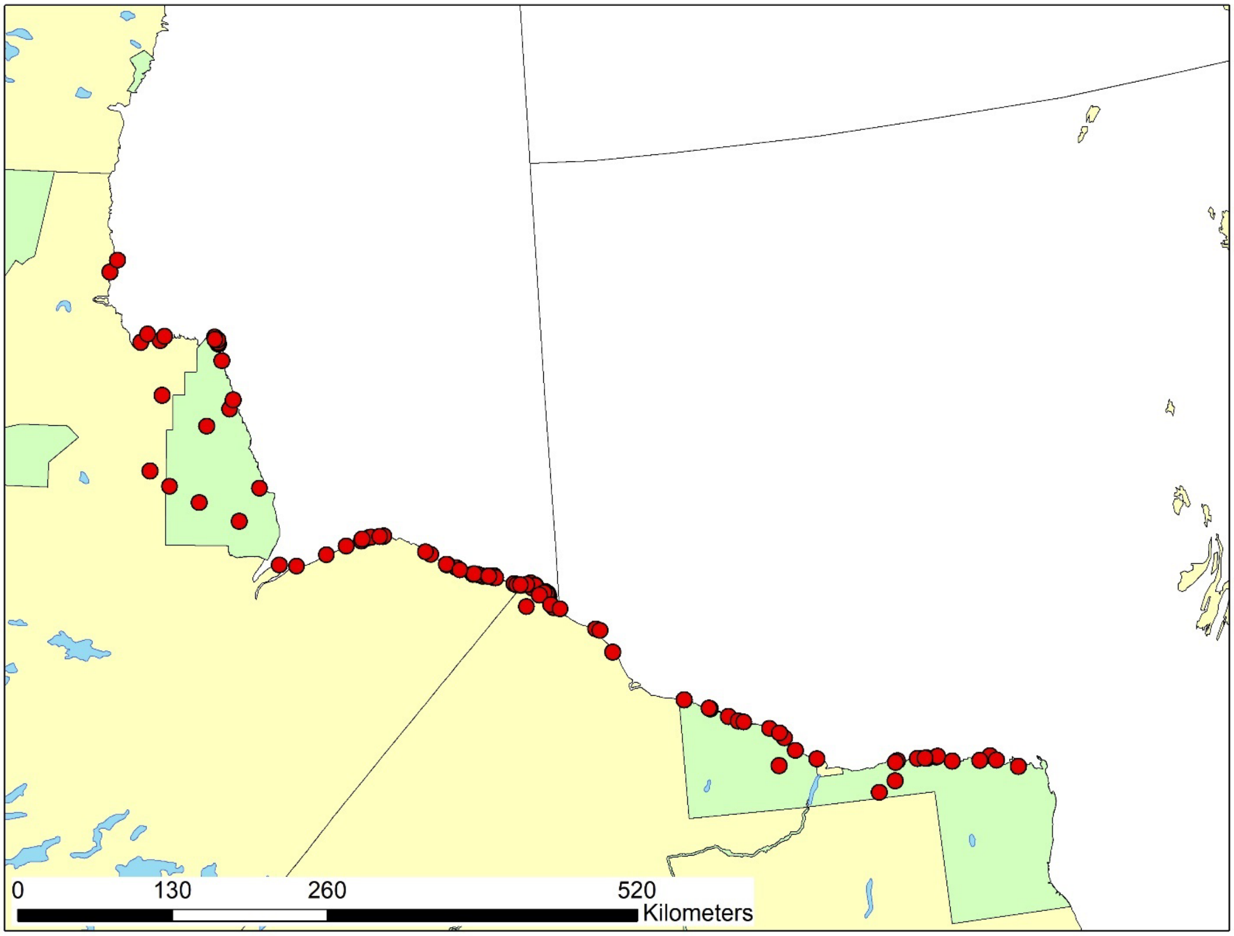
ᓴᕿᔮᖅᑐᖅ 5. ᓇᓃᓐᓂᖏᑦ ᓇᓄᐃᑦ ᓂᕿᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ 2021-ᒥ ᑕᐃᑲᓂ ᓄᓇ 2 ᐅᑯᐊᓗ ᐊᔾᔨᖏᑦ ᐊᑐᓂ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓯᒪᔪᖅ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᒥ (ᐊᐅᐸᖅᑐᑦ ᐊᖕᒪᓗᖅᑐᑦ) ᑕᒪᒃᑮ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᖁᖅᓱᖅᑐᑦ ᐊᖕᒪᓗᖅᑐᑦ) ᐊᒻᒪᓗ ᓯᕗᓪᓕᖅᐹᒥᒃ ᐱᔭᐅᔭᖑᐅᖅᑐᑦ (ᕿᓂᖅᑐᑦ ᐊᖕᒪᓗᖅᑐᑦ).



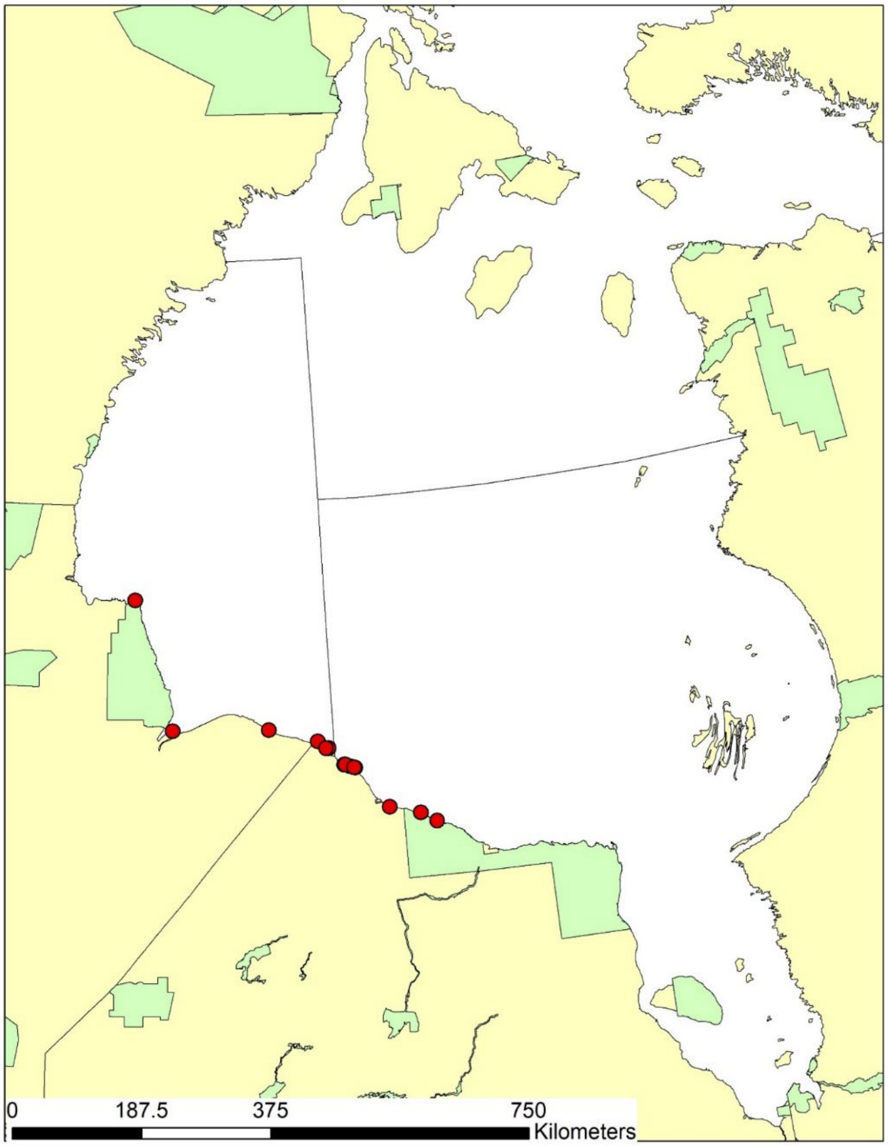
ᓴᖅᑭᔮᖅᑐᖅ 6. ᑭᖑᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᐃᓂᖏᑦ 96 ᓇᓄᐃᑦ (ᓇᓐᓄᒃᑕᐅᔪᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓂᕿᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᑖᒃᑯᓇᖓᑦ ᓴᖅᑭᔮᖅᑐᑦ 5) ᐅᑯᐊᓗ ᓂᕿᖏᑦ ᖃᐅᔨᕈᑎᑖᕐᕕᐅᓚᐅᖅᑐᖅ ᑕᐃᑲᓂ ᔫᓇ 2 2021-ᒥ.



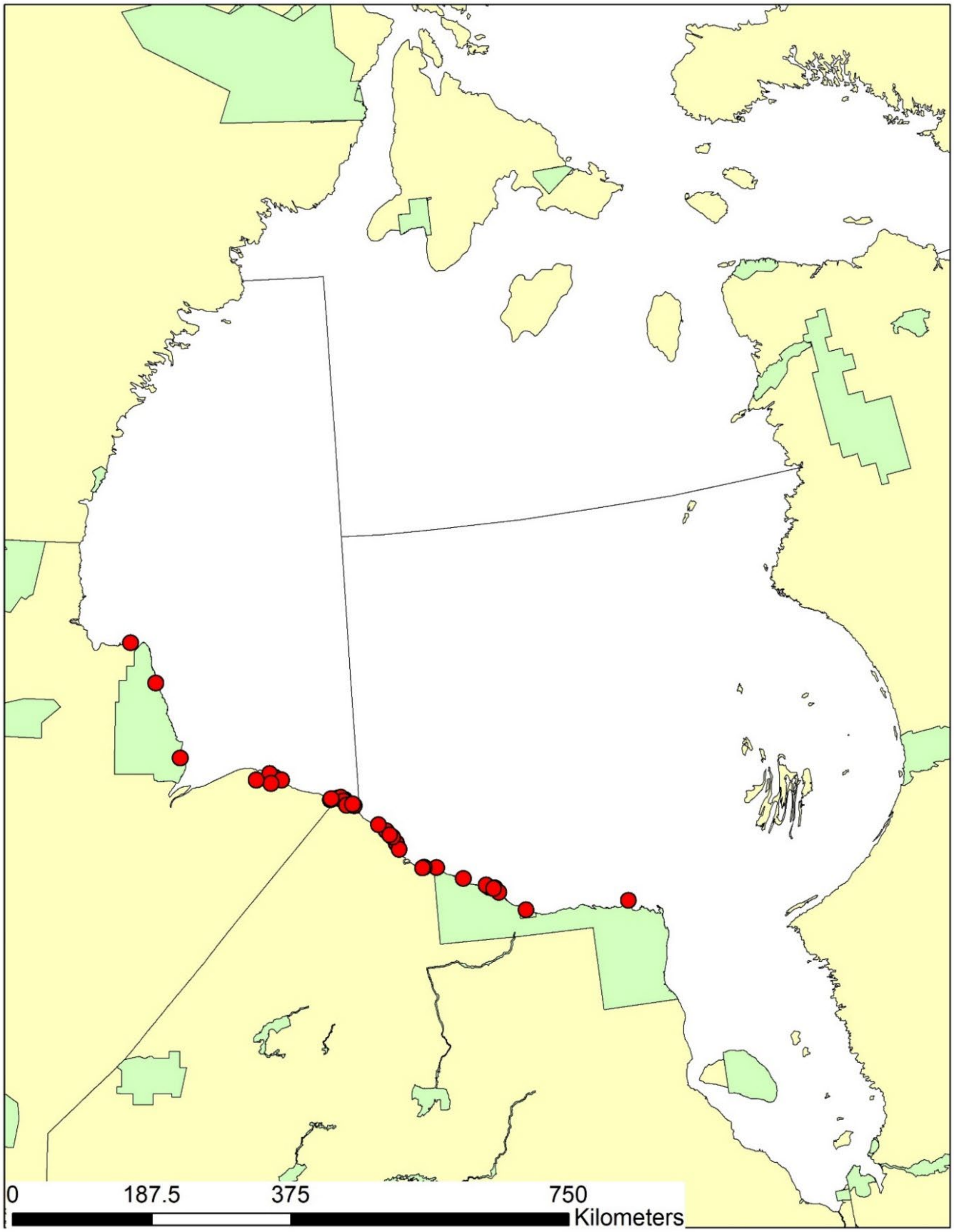
ᓴᕐᑭᔮᖅᑐᖅ 7. ᕐᑲᐅᔨᓴᕈᑎᑖᕐᕙᐅᔪᑦ ᐃᓂᖏᑦ 65 ᓇᓄᐃᑦ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᐅᑯᐊᓗ ᕐᑲᐅᔨᓴᕈᑎᑖᕐᕙᐅᓚᐅᖅᑐᑦ ᑭᖑᓂᑦᑎᓐᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ (ᐊᐅᐸᖅᑐᑦ ᐊᖕᒪᓗᖅᑐᑦ), ᑕᒪᒃᑮ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᐊᒻᒪᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᕐᑯᖅᓯᖅᑐᑦ ᐊᖕᒪᓗᖅᑐᑦ), ᐊᒻᒪᓗ ᑖᒃᑯᐊᑐᐊᖅ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᐆᔭᐅᔭᑦ ᐊᖕᒪᓗᖅᑐᑦ).



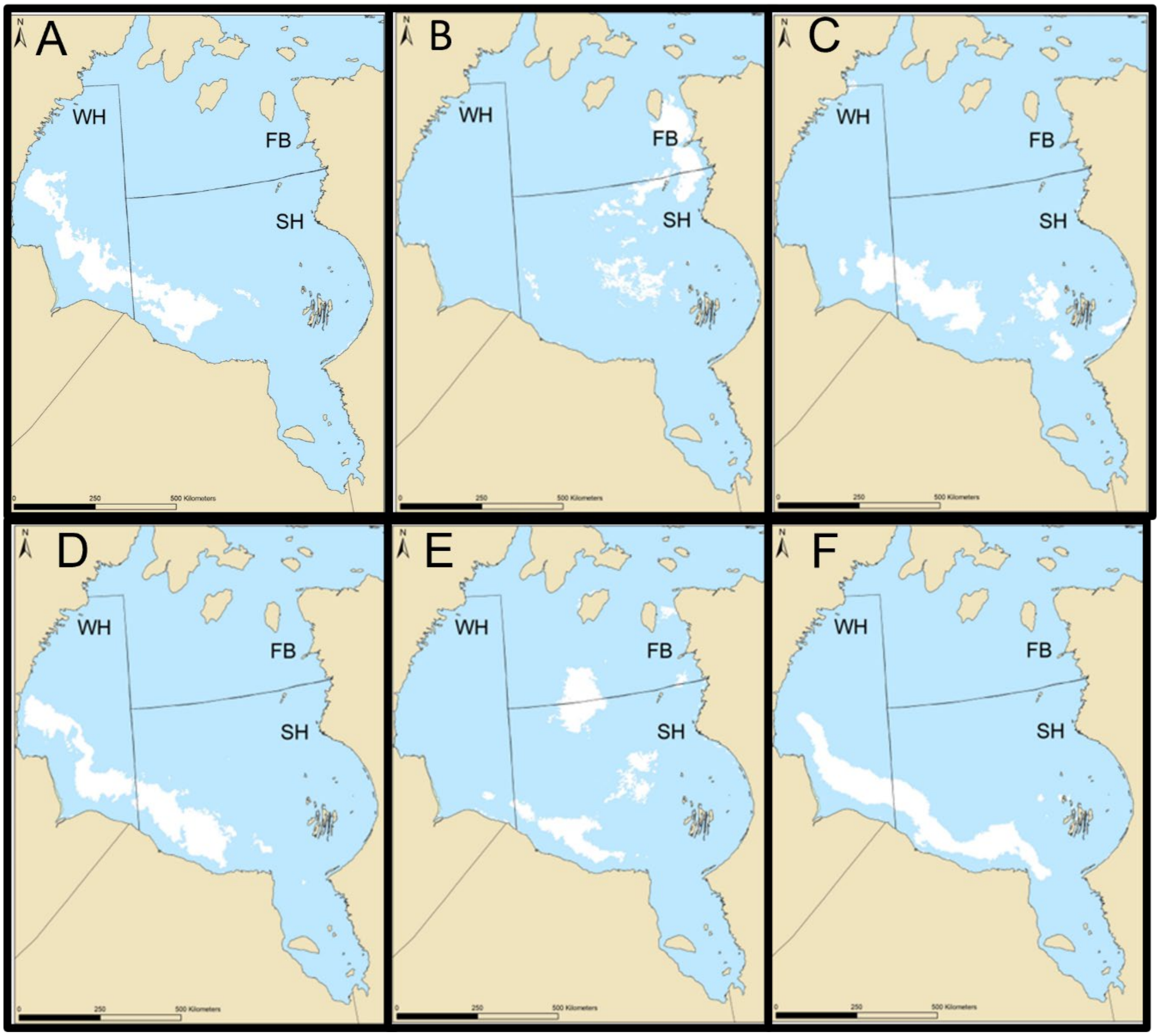
ᓴᖅᑭᔮᖅᑐᖅ 8. ᑭᙳᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᐃᓂᖏᑦ 65 ᓇᓄᐃᑦ (ᓴᖅᑭᔮᖅᑐᖅ 7) ᐅᑯᐊᓗ ᓂᕿᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᓯᒪᔪᑦ 2021-ᒥ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ.



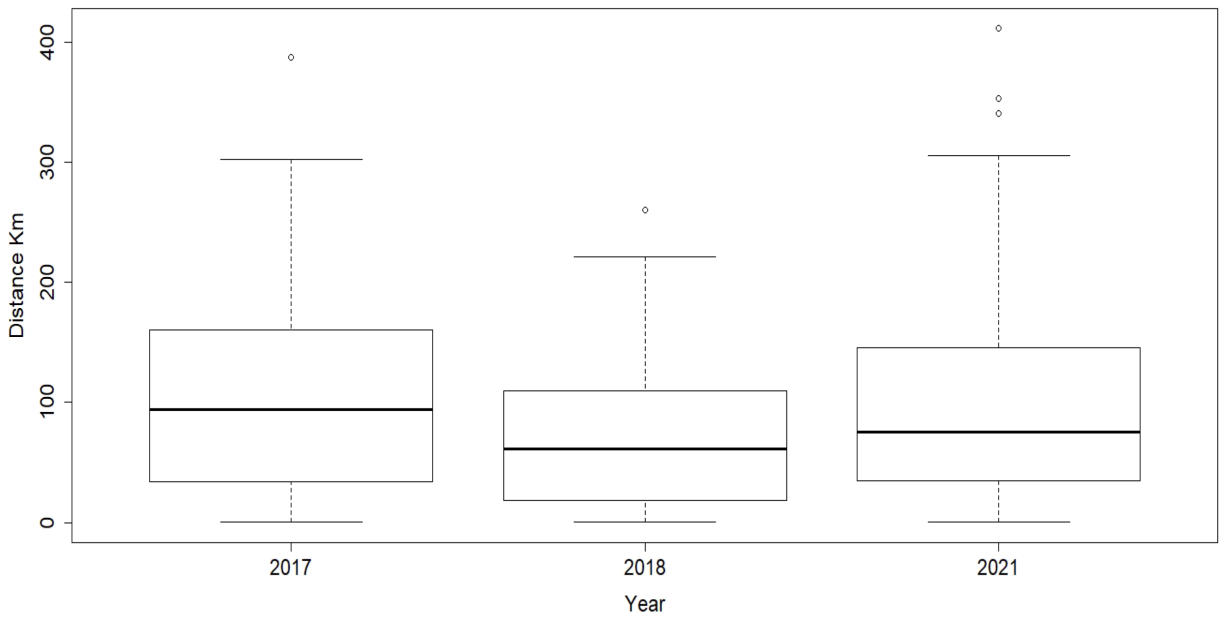
ᓴᕿᔮᖅᑐᖅ 9. ᓇᒧᖕᖓᓯᒪᓂᖏᑦ ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ 2021-ᒥ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᓚᐅᖅᑐᑦ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᑕᐃᑲᓂ 2022 (ᓲᕐᓗ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᓇᓖᓐᓂᖏᑦ 2021-ᒥ ᑕᐃᒪᓗ ᑕᐃᑲᓂ ᓂᕆᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓖᓐᓂᖏᑦ 2022-ᒥ ᑖᒃᑯᐊᑦᑕᐃᓐᓇᐃᑦ ᓇᓄᐃᑦ).



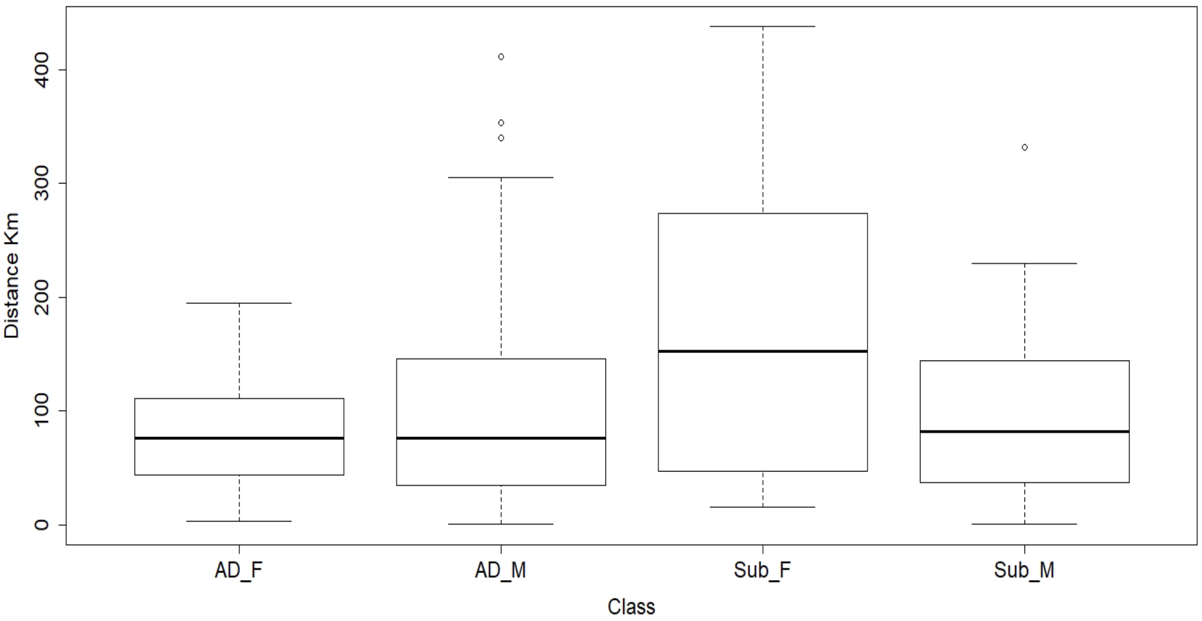
ᓴᕿᔮᖅᑐᖅ 10. ᓇᒧᙵᓯᒪᓂᖏᑦ ᓇᓄᐃᑦ ᑕᐃᑲᓂ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ 2021-ᒥ ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᒃᑲᓐᓂᓚᐅᖅᑐᑦ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᕐᒥ ᑕᐃᑲᓂ 2022 (ᓘᕐᓗ ᓂᒋᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓅᓐᓂᖏᑦ 2021-ᒥ ᑕᐃᒪᓗ ᑕᐃᑲᓂ ᐊᑭᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ ᓇᓅᓐᓂᖏᑦ 2022-ᒥ ᑖᒃᑯᐊᑦᑕᐃᓐᓇᐃᑦ ᓇᓄᐃᑦ).



ᓴᕿᔮᖅᑐᖅ 11. ᓯᑯᒐᓛᖕᓂᖓ (ᑕᐳᓯᖅ ᓯᑯᐊ ᑕᒪᕐᒥᒃ ≥ 30% ᑕᐃᑲᓂ ᑕᓇᐅᖅ ᓯᑯᐊ ᐊᖏᓂᖃᖅᑎᓪᓗᒍ 10% ᐅᑭᐅᒃᑯᑦ ᑕᒪᒃᑭᖅᓯᒪᓪᓗᓂ) ᑕᐃᑲᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ (ᐱᖃᓯᐅᑎᓇᒥ ᔭᐃᒻᔅ ᐸᐃ) ᑕᐃᑲᓂ 2017 (A), 2018 (B), 2019 (C), 2020 (D), 2021 (E) ᐊᒻᒪᓗ 2022 (F).



ᓴᕿᔮᖅᑐᖅ 12. ᑭᑉᐹᓇᒃᑑᑎᑎᒍᑦ ᑎᑎᕋᖅᓯᒪᔪᓪᓗ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᓯᖕᓂᖓ (ᑭᓚᒦᑕᔅ) ᐃᓐᓇᕐᓄᑦ ᐊᖑᑎᓄᑦ ᖃᐅᔮᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᐊᕐᖀᒍᑕᒫᑦ 2017 (2017/2018, n=32), 2018 (2018/2019, n=80) ᐊᒻᒪ 2021 (2021/2022, n=113) ᑕᐃᑲᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᐃᓗᐊᓃᑦᑐᖅ ᑭᒡᒐᖅᑐᖅᑐᖅ 25% ᐊᒻᒪ 75%, ᑐᑭᓕᐊᑦᑎᐊᖅᑐᑦ ᖅ ᑭᒡᒐᖅᑐᖅᑐᑦ ᐱᖃᑖᓂᒃ ᐊᒻᒪ ᒥᑭᔫᑎᓂ ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᖃᓂᒋᔭᖕᒐᓃᑦᑐᑦ ᐊᖕᒪᓗᖅᑑᑎᑯᓗᐃᑦ ᑭᒡᒐᖅᑐᖅᑐᑦ ᓯᓚᑖᓂᒃ.



ᓴᕿᔮᖅᑐᖅ 13. ᑭᐸᕝᓇᒃᑑᑎᑎᒍᑦ ᑎᑎᕋᖅᓯᒪᔪᓪᓗ ᓇᔪᖅᐸᒃᑕᒥᖕᓃᓐᖏᑦᑐᑦ ᐅᖓᓯᖕᓂᖓ (ᑭᓚᒦᑕᕐ) ᐃᓐᓇᕐᓄᑦ ᐊᕐᓇᑦ (ᐃᓐᓇᖅ_ᐊᕐᓇᖅ, n = 27), ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ (ᐃᓐᓇᖅ_ᐊᖑᑦ, n=113), ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᕐᓇᐃᑦ (ᐃᓐᓇᕈᓕᖅᑐᖅ_ᐊᕐᓇᖅ, n=20), ᐃᓐᓇᕈᓕᖅᑐᑦ ᐊᖑᑎᑦ (ᐃᓐᓇᕈᓕᖅᑐᑦ _ᐊᖑᑎᑦ, n=29) ᖃᐅᔨᓴᕈᑎᑖᕐᕕᐅᔪᑦ ᐊᑖᓂ ᐋᒍᓯ/ᓯᑎᐱᕆ 2021 ᐊᒻᒪᓗ 2022 ᑕᓯᐅᔭᕐᔪᐊᕐᒥ. ᑖᒃᑯᐊ ᑎᓴᒪᑦ ᐃᓗᐊᓂ ᐱᓯᒪᓂᖃᖅᑐᑦ 25% ᐊᒻᒪᓗ 75% ᑎᑎᑯᑖᒃᓯᒪᔪᑦ ᑮᒃᑲᖅᑐᖅᑐᖅ ᐱᖃᑖᓂᒃ ᐊᒻᒪᓗ ᒥᑭᔫᑎᓂ ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᖃᓂᒋᔭᖓᑳᓃᑦᑐᑦ ᐊᖕᒪᓗᖅᑑᑎᑯᓗᐃᑦ ᑮᒃᑲᖅᑐᖅᑐᑦ ᓯᓚᑖᓂᒃ.

ᐅᑯᓇᙵ ᑐᓂᔭᐅᔪᖅ

ᓄᓇᕗᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ

ᐅᑯᓇᙵ

ᑐᓴᖅᑕᐅᔪᒃᓴᓄᑦ: **X** **ᐃᓱᒪᓕᐊᕐᒍᔭᖅ:**

ᐱᔾᔪᑕᐅᔪᖅ: ᖃᐅᔨᓴᕈᑎᖃᕈᑦ 2021-ᒥ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓂᒃ ᓄᓇᓕᓐᓄᓪᓗ ᑐᓴᕆᐊᕐᕕᖃᕐᓂᕐᒧᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ (SH) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦᑕ.

ᐱᓕᕆᐊᕐᒍᓯᒪᔪᑦ:

- ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ (SH) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦᑕ ᐱᖅᑲᑎᒌᒍᑕᐅᔪᖅ ᐋᓐᑎᐅᕆᐅ, ᑯᐯᒃ, ᐊᒻᒪ ᓄᓇᕗᑦ (ᐊᔾᔨᙳᐊᖅ 1). ᑭᖑᓪᓕᖅᐹᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓇᓚᐅᑦᓵᖅᑕᐅᓯᒪᔪᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᐱᓕᕆᐊᕐᒍᓚᐅᖅᐳᑦ 2011-ᒥ ᐊᒻᒪ 2016-ᒥ, 943 ᐊᒻᒪ 780-ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ ᓇᓄᐃᑦ. ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦᑕ ᖃᑦᔩᓐᓇᕈᖅᐸᓪᓕᐊᓂᕋᖅᑕᐅᓚᐅᖅᑐᑦ ᑕᐃᑦᓱᒪᓂ.

- ᓴᓂᑭᓗᐊᕐᒥ ᐊᕐᖁᓇᓱᓲᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ. ᒫᓇᐅᔪᖅ ᑲᑎᓪᓗᒋᑦ ᐊᕐᖁᓇᓱᒃᑕᐅᔫᓇᖅᑐᑦ (TAH) ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 25–ᖑᕗᑦ ᓇᓄᐃᑦ ᐊᕐᕎᒍᑕᒫᑦ. ᑖᒃᑯᐊ ᑲᑎᓪᓗᒋᑦ ᐊᕐᖁᓇᓱᒃᑕᐅᔫᓇᖅᑐᑦ ᐋᖅᑭᑦᑕᐅᓚᐅᖅᐳᖅ ᐊᓯᔾᔨᒋᐊᕋᑦᓴᐅᓪᓗᓂ ᖃᑦᓯᕿᑭᓐᓇᕐᓂᕆᔭᐅᔪᑎᒍᑦ ᐊᑐᖅᑕᐅᕙᑦᑐᑎᒍᑦ (FQS) 2:1–ᒥ ᐊᕐᖁᓇᓱᑦᑕᐅᔪᖅ ᑭᓲᓂᖏᑕ ᖃᑦᓯᐅᓂᖏᓐᓄᑦ ᑲᔪᓯᑎᑕᐅᓚᐅᖅᓯᒪᓪᓗᓂᓗ 1:1–ᒥ ᐊᕐᖑᔭᐅᔪᑦ ᑭᓲᓂᖏᑕ ᖃᑦᓯᐅᓂᖏᓐᓄᑦ ᓄᑖᖅ ᐊᕐᖁᓇᓱᑦᑕᐅᔪᓄᑦ ᐊᐅᓚᑦᓯᕙᓐᓂᕐᒧᑦ ᐊᒥᐊᒃᑯᓯᔭᐅᔪᓄᓪᓗ ᖃᑦᓯᐅᓂᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᕙᒃᑐᖅ (HACCS) ᐊᑐᓕᓚᐅᖅᑎᓪᓗᒍ 2022–ᒥ. ᓇᓄᒃᑕᐅᕙᔪᒃᑐᑦ 2011/2012 ᐊᒻᒪ 2022/2023 ᐊᕐᖁᓇᓱᓐᓇᐅᑎᓪᓗᒍ 28–ᖑᓚᐅᖅᐳᑦ ᓇᓄᐃᑦ ᐊᕐᕎᒍᑕᒫᕐᒧᑦ (ᐊᔾᔨᙳᐊᖅ 2).

- ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᒻᒪ ᐱᕙᓐᓇᖅᐸᓯᕙᓂ ᑕᓯᐅᔭᕐᔪᐊᑉ (WH) ᒪᓕᒃᑦᑕᖅᓯᒪᕗᑦ ᐊᕐᕎᒍᓄᑦ ᑕᓪᓕᒪᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ, ᖃᐅᔨᓴᖅᑕᐅᕋᑖᓚᐅᖅᑐᖅ ᐱᔭᕇᖅᑕᐅᑦᓱᓂ 2021–ᒥ. ᓄᓇᓕᓐᓂ ᑐᓴᕆᐊᕐᕕᖃᓚᐅᖅᑎᓪᓗᒋᑦ ᐅᖃᓚᐅᑎᒃᑯᑦ 2021–ᒥ (ᐅᐸᒍᑎᓪᓗᑎᒃ ᑐᓴᕆᐊᕐᕕᖃᕈᓐᓇᓚᐅᕐᒋᒻᒪᑕ ᓄᕙᕐᔪᐊᕐᓇᖅ-19–ᒧ ᐊᐅᓪᓚᑲᑕᓗᐊᕆᐊᖄᒋᑎᓪᓗᒋᑦ), ᖃᐅᔨᓴᖅᑕᐅᓯᒋᐊᓚᐅᖅᐳᖅ ᐊᐅᒡᒍᓯ 2021–ᒥ ᐱᔭᕇᖅᓱᑎᓪᓗ ᓯᑦᑎᒻᕙ 2021–ᒥ. ᖃᕆᑕᔫᒃᑯᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕈᑎᒥᒃ ᐊᑐᓚᐅᖅᐳᑦ ᑖᑦᓱᒥᙶ ᖃᐅᔨᓴᖅᑎᓪᓗᒋᑦ. ᐱᙳᓇᖅᐸᓯᙶᓂ ᑕᓯᐅᔭᕐᔪᐊᑉ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᖅ ᐊᑕᐅᑦᓯᒃᑯᑦ ᓄᓇᕗᒥ ᒐᕙᒪᒃᑯᓐᓄᑦ.

- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ (ECCC) ᖃᐅᔨᓴᖃᑦᑕᖅᓯᒪᔪᑦ ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ 1980–ᒥᓂᑦ. ᖃᐅᔨᓴᖃᑦᑕᐃᓐᓇᖅᑐᑦ ᐃᒪᐃᓕᐅᖅᓯᑎᒃ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᓪᓗᑎᒃ ᓇᐅᒃᑲᖅᑕᐅᒻᒧᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂᒃ ᐱᓕᕆᐊᖑᓯᒪᓕᖅᑐᑦ 2017–ᒥ ᐱᒋᐊᓚᐅᖅᓯᒪᓪᓗᓂ ᒫᓂᑑᐸᒥ ᐊᒻᒪ ᐋᓐᑎᐅᕆᐅ ᓯᒡᔭᖓᓂ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᖅᑐᑦ ᓇᒧᙵᐅᕙᓪᓕᐊᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᖃᑦᑕᓚᐅᖅᐳᑦ ᐅᑯᓇᓂ, ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ/ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᒻᒪᓗ ᓯᑯᐃᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᓐᓂᒃ ᑕᓯᐅᔾᔪᐊᒥ. ᖃᐅᔨᓵᕈᑕᐅᔪᑦ ᑕᕝᕙᙵᑦ ᐅᓂᒃᑳᒥ ᑐᑭᓯᓇᑦᓯᐊᓚᐅᖅᐳᑦ ᐃᑲᔫᑕᐅᓪᓗᑎᓪᓗ ᑐᑭᓯᓇᖅᓯᑎᓪᓗᒋᑦ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᖅᓯᒪᔪᑦ ᓇᓚᐅᑦᓵᖅᑕᐅᓯᒪᔪᑦ 2021–ᒥ ᓄᓇᕗᑦ ᑲᕙᒪᒃᑯᑦ ᖃᖓᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ.

ᒫᓐᓇ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ:

- ᑭᖑᓪᓕᖅᐹᖅ ᐅᓂᒃᑳᖅ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓪᓗ ᖃᐅᔨᓴᕈᑕᐅᔪᖅ 2021–ᒥ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᑐᓂᐅᖅᑲᐅᑕᐅᓚᐅᖅᐳᑦ ᐊᐅᓚᑦᑎᔨᐅᖃᑕᐅᔪᓕᒫᓄᑦ ᓅᕙᐃᒻᕙ 2022–ᒥ. ᓄᑖᑦ ᐊᒥᓲᓂᖏᑦ ᒥᒃᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1003 ᓇᓄᐃᑦ ᐊᒥᓲᓂᖅᓴᐅᔪᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ 2011-ᒥ ᐊᒻᒪ 2016-ᒥ.

- ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᐊᓂᓗ ᓇᓚᐅᑦᓵᖅᑕᐅᔪᑦ 2011, 2016, ᐊᒻᒪ 2021 ᑲᑎᑕᐅᓚᐅᖅᐳᑦ ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᐅᓂᒃᑳᓕᐊᒥ. ᑕᒪᓐᓇ ᓴᖅᑭᑦᓯᓚᐅᖅᐳᖅ ᑲᑎᓪᓗᒋᑦ ᓇᓚᐅᑦᓵᖅᑕᐅᔪᓂᑦ 1892, 1622, ᐊᒻᒪ 1621 ᓇᓄᕐᓂᒃ, ᐊᑐᓂ. ᑕᒪᕐᒥᒃ ᐅᓄᕐᓂᖏᑦᑕ ᑲᑎᓯᒪᑎᓪᓗᒋᑦ, ᑕᑯᒃᓴᐅᑎᑦᓯᔪᖅ ᓱᕐᖁᓗᐊᙱᑦᑐᓂᒃ ᐅᓄᕐᓂᖏᓐᓂᒃ (ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᐊᒻᒪ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ) 2016 ᐊᒻᒪ 2021 ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ.

- ᑭᓪᓕᖃᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᖃᖓᑕᔫᒃᑯᑦ ᓈᓴᐃᕙᒃᑐᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ, ᓇᓗᓇᓚᐅᖅᑐᖅ ᖃᓄᑎᒋ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᖅᑎᒋᒻᒪᖔᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 2021–ᒥ ᓇᓄᐃᑦ ᓅᐸᓪᓕᐊᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ, ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᐋᓐᑎᐅᕆᐅᒥᓪᓗ ᓇᑭᙶᖕᓂᖏᓐᓂᑦ ᓇᓗᓇᐃᒃᑯᑦᑎᖅᓯᐃᓃᑦ ᐱᔭᐅᒃᑲᓐᓂᖅᐸᒃᑐᓪᓗ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᖅᑕᐅᓪᓗᑎᒃ.

- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᑲᑕᒃᑐᑦ ᐅᓂᒃᑳᖓᓂ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐅᓄᖅᓯᓯᒪᓂᖏᓐᓄᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 2021–ᖑᑎᓪᓗᒍ ᓇᓄᐃᑦ ᓅᐸᓪᓕᐊᓂᖏᓐᓄᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᐊᓂ, ᓯᑯᖃᕐᓂᖓᓄᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᐊᓂ ᖃᖓᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᓗᒍ.

- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᑲᑕᒃᑐᑦ ᐅᓂᒃᑳᖓᓂ ᖃᐅᔨᓴᓚᐅᕐᒥᔪᑦ ᓇᓐᓄᕝᕕᐅᔪᕕᓂᕐᓂ. ᓇᓗᓇᐃᖅᓯᒃᑲᓐᓂᖅᐳᖅ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᐅᓚᑦᓯᔩᑦ ᑭᓪᓕᒋᔭᖓ ᐊᑐᕈᓐᓇᕐᓂᖓᓂᒃ ᓇᓄᑦᑕᐅᔪᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓯᕗᓪᓕᕐᒥ ᑎᒥᖓᓂ ᐲᖅᓯᕝᕕᐅᓚᐅᕐᓂᖏᓐᓄᑦ ᑕᓯᐅᔾᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ, ᓇᓐᓄᑦᑕᐅᔪᑦ ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᓇᓐᓄᒃᑕᐅᔪᑦ ᓯᕗᓪᓕᖅᐹᒥ ᑎᒥᖓᓂ ᐲᖅᓯᕝᕕᐅᓚᐅᖅᓯᑎᒃ ᑕᓯᐅᔾᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ.

ᑐᓴᕆᐊᕐᕕᖅᑲᕐᓃᑦ:

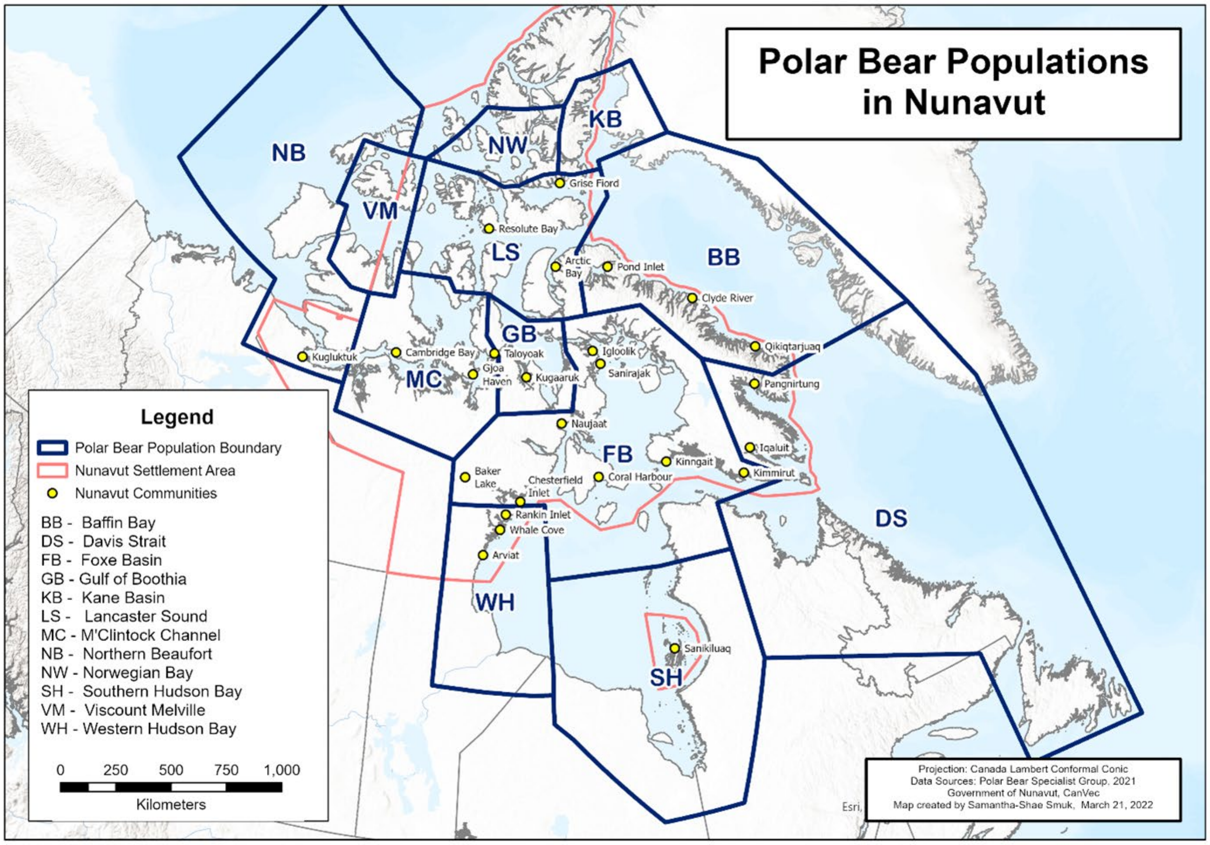
ᐅᐸᒍᑎᓪᓗᑎᒃ ᓄᓇᓕᖕᓂ ᑐᓴᕆᐊᕐᕕᖅᑲᕐᓂᕐᒧᑦ ᓴᓂᑭᓗᐊᕐᒥ ᐆᒪᔪᕐᓂᐊᖅᑎᒃᑯᖕᓂᒃ (HTA) ᒪᐃ 13, 2024−ᒥ. ᑐᓴᕆᐊᕐᕕᖅᑲᕐᓂᐅᔪᑦ ᑐᕌᒐᖅᑲᓗᐊᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᔾᔪᑎᓂᒃ ᖃᖓᑕᔫᒃᑯᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᓈᓴᐃᑎᓪᓗᒋᑦ ᐊᒻᒪᓗ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑎᒥᖓᓂ ᐲᖅᓯᓪᓗᑎᒃ ᖃᐅᔨᓴᖅᑐᓂᑦ. ᐆᒪᔪᕐᓂᐊᖅᑎᒃᑯᑦ ᐊᒥᓱᓂᑦ ᐊᐱᖅᑯᑎᒃᓴᖅᑲᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᐅᑉ ᒥᑦᓵᓄᑦ ᖃᓄᐃᙱᑦᓯᐊᖕᓂᖏᓐᓂᓪᓗ ᓇᓄᐃᑦ ᐸᕝᕕᓴᓐᓂᕐᓴᐅᓪᓗᑎᒃ ᖃᐅᔨᓴᕐᓂᕐᓂᑦ (ᕐᑯᖓᓯᕈᖅᓯᓂᖅ ᑎᒥᒥᓪᓗ ᐲᖅᓯᓂᖅ). ᐆᒪᔪᕐᓂᐊᖅᑎᒃᑯᑦ ᐃᓱᒪᒋᔭᖃᓚᐅᖅᑐᑦ ᑐᓂᓯᔪᒪᓪᓗᑎᒃ ᓇᒻᒥᓂᖅ ᑎᑎᖅᑲᒥᓂᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐃᓱᒪᒋᔭᖏᓐᓂᑦ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᕐᑯᔭᓂᑦ. ᐃᖅᑲᓇᐃᔭᖅᑏᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂᑦ ᑲᑎᒪᖅᑕᐅᓚᐅᖅᐳᑦ. ᑕᑯᓪᓗᒋᑦ ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᑐᓴᕆᐊᕐᕕᖅᑲᕐᓂᕐᒧᑦ ᓇᐃᓈᖅᓯᒪᔪᑦ ᐅᓂᒃᑳᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᖏᓐᓂᑦ.

ᐊᑐᓕᕐᑯᔭᐅᔪᑦ:

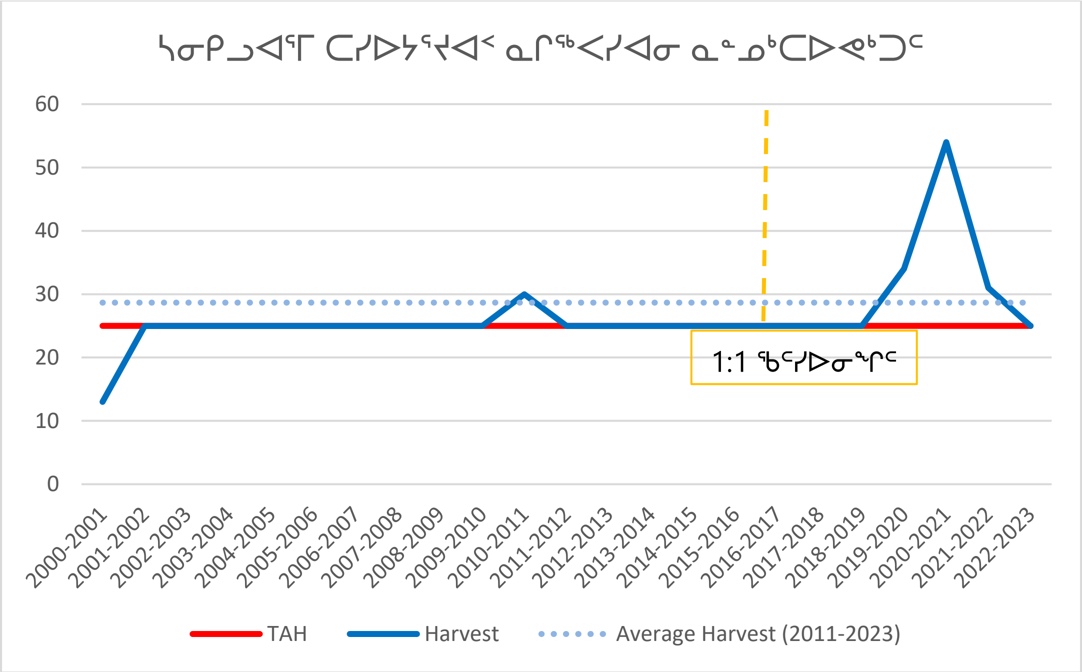
1. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ **ᐊᓯᔾᔩᕐᑯᔨᙱᑦᑐᑦ ᒫᓐᓇ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ 25−ᖑᓪᓗᑎᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐅᓄᕐᓂᖏᑦᑕ**. ᐃᓱᒪᒋᓪᓗᒋᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᒻᒪ ᐱᖓᓐᓇᖅᐸᓯᐊᓂ ᓇᓚᐅᑦᑖᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᓪᓗᒋᑦ 2016 ᐊᒻᒪ 2021 ᐊᑯᓐᓂᖓᓂ, ᓇᓄᐃᑦ ᓱᕐᕋᓗᐊᖅᓯᒪᙱᑦᑐᑦ ᒫᓐᓇ ᐊᕙᑎᖏᑦ ᖃᓄᐃᓕᖓᓂᖏ ᐊᖑᓇᓱᒃᑕᐅᕙᒃᑐᓪᓗ ᒪᓕᓪᓗᒋᑦ.

2. ᒫᓐᓇ ᑲᓇᑕᓕᒫᒥ ᓄᓇᕐᔪᐊᓕᒫᒥᓗ ᓇᓄᐃᑦ ᐊᐅᓚᑕᐅᔪᑦ ᐱᓕᕆᐊᖑᕙᓪᓕᐊᔪᑦ ᒪᓕᒃᓗᒍ, ᐊᕙᑎᔨᕆᔨᒃᑯᑦ ᑐᒃᓯᕋᖅᐳᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐋᖅᑭᒃᓯᐃᕐᑯᓪᓗᒋᑦ ᐊᐅᓚᑦᓯᓂᕐᒧᑦ ᑐᕌᕆᔭᐅᔪᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᖏᓐᓄᑦ. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐃᒪᐃᕐᑯᔨᕗᑦ ᐊᐅᓚᑦᓯᓂᕐᒧᑦ ᑐᕌᕆᔭᐅᔪᑦ **ᒫᓐᓇ ᐅᓄᕐᓂᖏᑦ ᑲᔪᓯᑎᓪᓗᒋᑦ**.

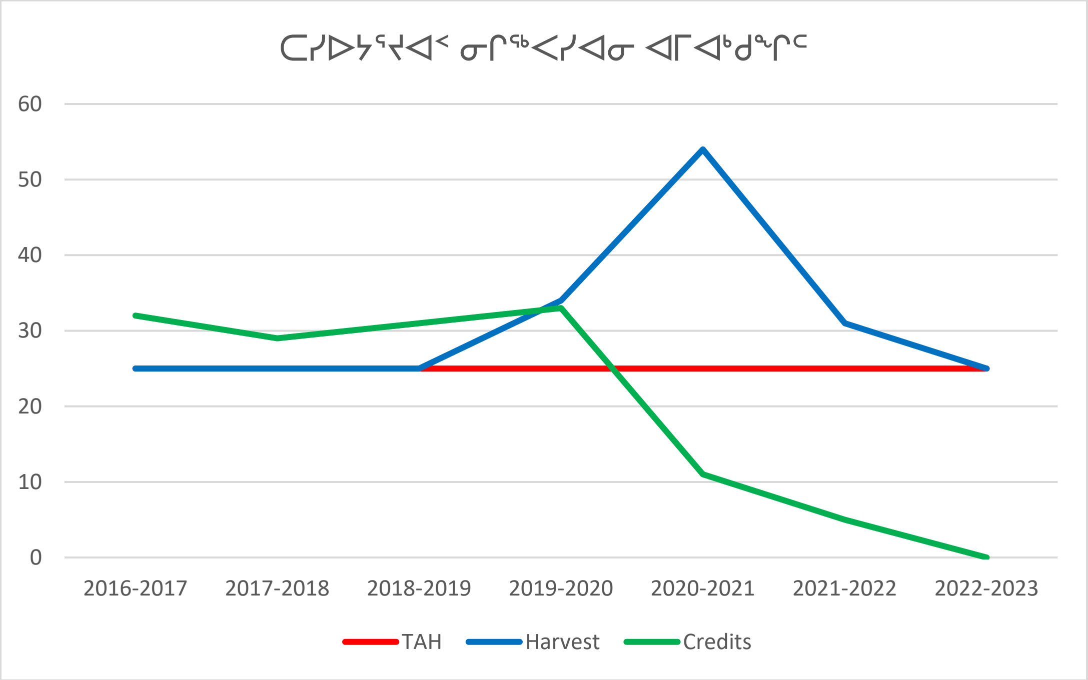
ᐅᐃᒍᖓ 1



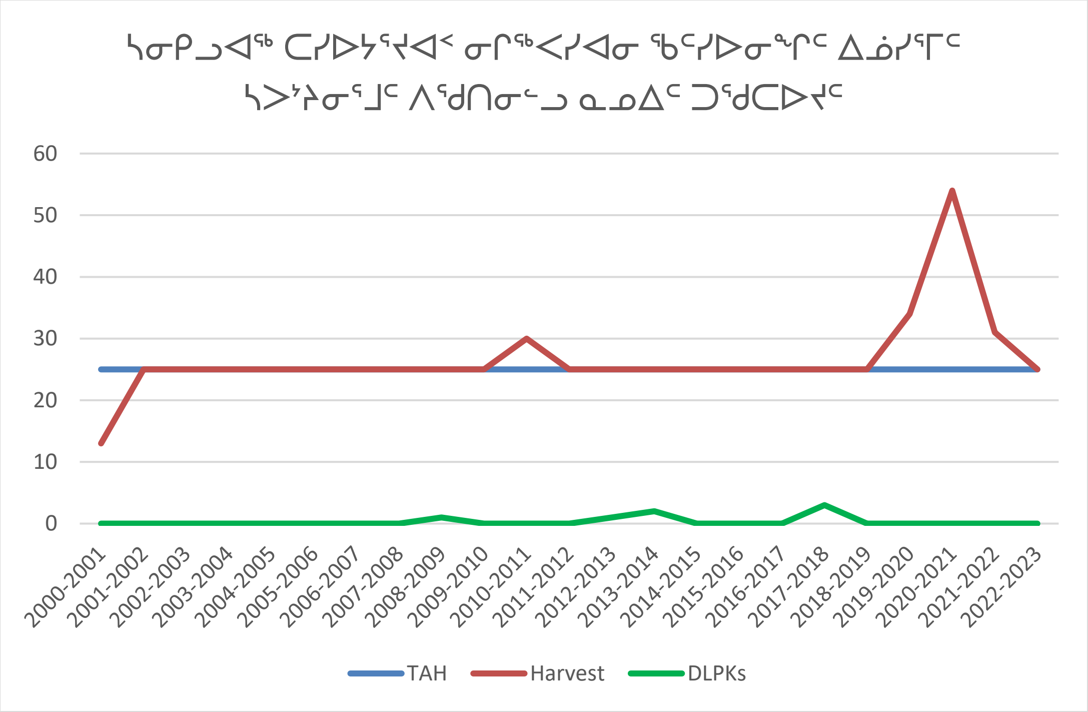
ᐊᔾᔨᙳᐊᖅ 1. ᖃᓄᐃᓕᖓᓂᖏᑦ ᓄᓇᕗᒻᒥ ᓇᓄᐃᑦ (WH = ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ, SH = ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ).



ᐊᔾᔨᙳᐊᖅ 2. ᖄᓄᐃᓕᖓᓂᖏᑦ ᑕᓯᐅᔾᕿᐊᑉ ᓂᔨᖅᐸᓯᐊᓂ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH), ᐊᑐᖅᑕᐅᓪᓚᕆᒃᑐᑦ ᐊᑐᖅᑕᐅᒐᔪᒃᑐᓪᓗ ᐊᖑᓇᑦᑕᐅᔪᓐᓇᖅᑐᑦ 2000–ᒥ ᐱᒋᐊᖅᓯᓂ.



ᐊᔾᔨᙳᐊᖅ 3. ᖄᓄᐃᓕᖓᓂᖏᑦ ᑕᓯᐅᔾᕿᐊᑉ ᓂᔨᖅᐸᓯᐊᓂ ᓇᓐᓄᒐᑦᓴᐃᑦ ᐊᒥᐊᒃᑯᖏᑦ 2016–ᒥ ᐱᒋᐊᖅᓯᓂ.



ᐊᔾᔨᙳᐊᖅ 4. ᖃᑦᓯᐅᓂᖏᑦ ᐃᓅᓯᕐᒥᑦ ᓴᐳᔾᔨᓂᕐᒧᑦ ᐱᕐᑯᑎᓂᓪᓗ ᓇᓄᐃᑦ ᑐᕐᑯᑕᐅᔪᑦ (DLPKs) ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 2000−ᒥ ᐱᒋᐊᖅᓯᓂ.

ᑲᑐᑎᓪᓗᒍ ᓄᓇᕗᑦ ᐱᕈᖅᐸᓪᓕᐊᑎᓪᓗᒍ
Building *Nunavut* Together
*Nunavu*liuqatigiingniq
Bâtir le *Nunavut* ensemble

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ
Department of Environment
Avatiliqiyikkut
Ministère de l’Environnement

ᓄᓇᓕᖕᓂᑦ ᑐᓴᕋᓱᖕᓃᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᕐᓂᒃ ᖃᖓᑕᓲᒃᑯᑦ ᓈᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ 2021−ᒥ

© Félix Boulanger

ᐱᒋᐊᕈᑎᒃᓯᑦ

ᐊᓖᓯ ᐳᕼᐋᕐᑦ
ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑎ
ᐊᕙᑎᓕᕆᔨᒃᑯᑦ
ᓄᓇᕗᑦ ᑲᕙᒪᖓ

ᔫ ᓄᐊᑦᕋᑉ
ᐊᕕᒃᑐᖅᓯᒪᔪᓂᑦ ᐆᒪᔪᕐᓂᐊᖅᑎ
ᐆᒪᔪᓂᒃ ᖃᐅᔨᓴᖅᑎᒃᑯᑦ ᐊᒻᒪ
ᖃᐅᔨᓴᖅᐸᓪᓕᐊᔨᑦ
ᐋᓐᑎᐅᓇᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓯᑦ ᐊᒻᒪ
ᓇᐹᖅᑐᓕᕆᔨ

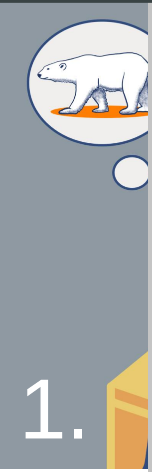
ᐊᓯᐊᓄᐊᕐᓂᖅ

- ᑕᐃᒪᖕᒐᑦ 2021-ᒥᑦ ᐊᓯᐊᖕᒍᖅᓯᒪᔪᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓂᑦ
  - ᐃᓅᔪᖕᓃᒃᑳᓚᓚᐅᖅᑐᖅ ᓇᓄᕐᓄᑦ ᐆᒪᔪᓕᕆᔨ ᒫᑲᔅ ᑕᐃᒃ ᐱᓂᕐᓗᐊᕿᓂᕐᒥᑦ
  - ᒪᖀᒃ ᓇᓄᕐᓄᑦ ᐆᒪᔪᓕᕆᔩᑦ ᓄᖅᑲᓚᐅᖅᑐᑦ ᐊᖀᓂ
  - ᒪᖀᒃ ᓄᓛᒃ ᓇᓄᕐᓄᑦ ᐆᒪᔪᓕᕆᔩᑦ ᑲᔪᓯᔪᑦ ᑐᖀᖕᒐᔪᒥᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1. ᓴᖅᑭᑦᑎᓗᑎᒃ ᖃᐅᔨᓴᕐᕕᒃᓴᒥᒃ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ



2. ᐅᖃᖃᑎᖃᕐᓗᑎᒃ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐋᖅᑭᒃᓯᒪᙱᓂᒃ ᑐᓴᕐᓗᑎᑦ ᐅᖃᐅᓯᐅᔪᓂᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1.

3. ᐃᓚᓕᐅᑎᒍᒋᑦ ᑭᐅᔾᔪᑕᐅᔪᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐋᖅᑭᒃᓯᒪᓂᕆᔭᐅᔪᒧᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

1.

4. ᖃᖕᒐᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓗᒋᑦ

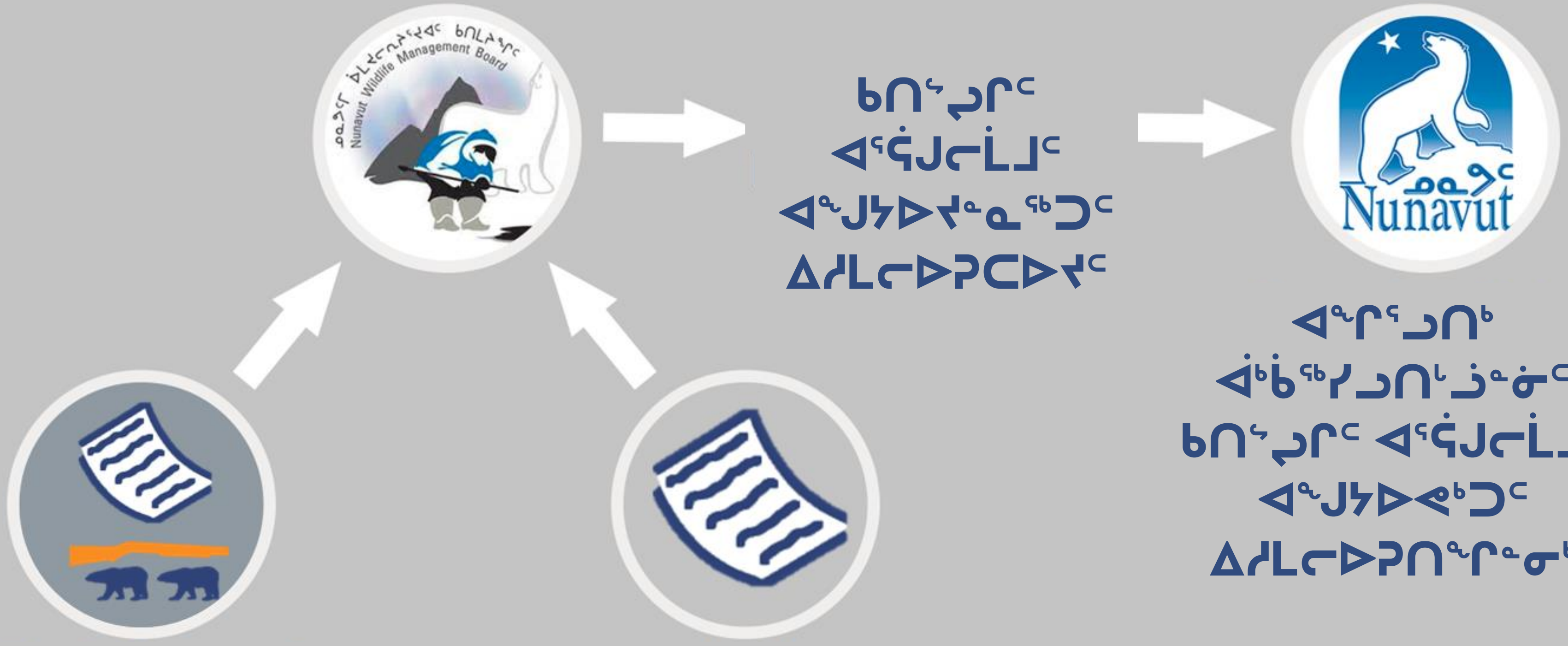
ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

5. ᕿᒥᕐᕈᓂᖅ ᐊᒻᒪ ᑎᑎᕋᕐᓂᖅ ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᒥᒃ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

6. ᑐᓴᕋᓱᒃᓂᖅ ᑭᖑᓪᓕᖅᐹᒥᑦ ᐅᓂᒃᑳᒥᑦ

ᖃᐅᔨᓴᖅᐸᓪᓚᐊᓂᖅ



ᒥᓂᔅᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ

ᑲᑎᔅᔪᒋᑦ
ᐊᖅᖂᒍᓕᒫᒧᑦ
ᐊᖑᔭᐅᔪᓐᓇᖅᑐᑦ
ᐃᓱᒪᓕᐅᕈᑕᐅᔪᑦ

ᐊᖏᕐᓗᑎᒃ
ᐋᒃᑳᖅᓯᓗᑎᒡᓘᓐᓃᑦ
ᑲᑎᔅᔪᒋᑦ ᐊᖅᖂᒍᓕᒫᒧᑦ
ᐊᖑᔭᐅᕙᒃᑐᑦ
ᐃᓱᒪᓕᐅᕈᑎᖏᓐᓂᒃ

ᑭᖑᓪᓕᖅᐹᖅ ᐅᓂᒃᑳᖅ ᐊᒻᒪ
ᑲᑎᓪᓗᒋᑦ ᐊᖅᖂᒍᓕᒫᖅᓯᐅᑎᑦ t
ᖃᔅᓯᐅᓂᖏᑦ
ᐊᑐᓕᖅᑯᔭᐅᔪᑦ

ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ
ᑭᖑᓪᓕᖅᐹᖅ ᐅᓂᒃᑳᖅ

7. ᑲᑎᔅᔪᒋᑦ ᐊᖑᔭᐅᔪᓐᓇᖅᑐᑦ ᐃᓱᒪᓕᐅᕈᑎᖏᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ

ᒥᓂᔅᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ

ᐊᖏᖅᑕᐅᔪᑦ

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ

ᐊᖏᕐᓗᑎᒃ ᐋᒃᑳᕐᓗᑎᒡᔫᓃᑦ ᑲᑎᕝᕙᕆᑦ ᐊᕐᕿᒍᓕᒫᒧᑦ ᐊᙳᓯᐅᕙᒃᑐᑦ ᐃᓱᒪᓕᐅᕈᑎᖏᓐᓂᒃ

ᐊᕚᑦᑐᖅᓯᒪᔪᓂ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔩᖏᑦ

ᐋᖅᑭᒃᓯᔪᑎᑦ ᖃᔅᓯᓂᒃ ᓂᕈᐊᖅᑕᐅᓂᒃ ᑕᒪᕐᒥᒃ ᓄᓇᓖᑦ ᐱᓂᐊᕐᒪᖔᑕ

ᓄᓇᓕᖕᓄᑦ ᓂᕈᐊᕐᕕᖃᖔᑦ ᓇᓯᐅᔾᔭᐅᔪᑦ

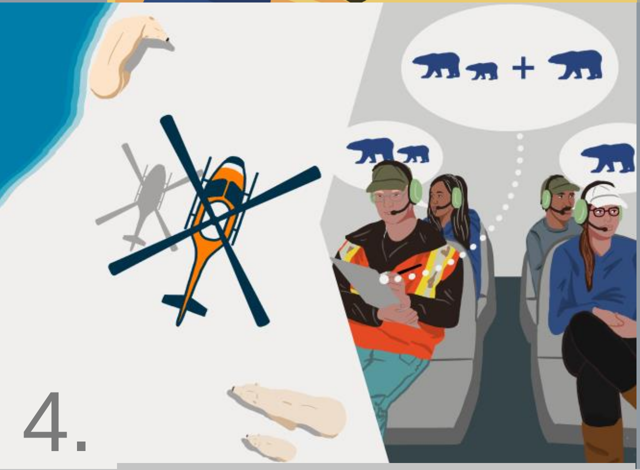
7. ᑲᑎᕝᕙᕆᑦ ᐊᙳᓯᐅᔪᓐᓇᖅᑐᑦ ᐃᓱᒪᓕᐅᕈᑎᖏᑦ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓂᖅ


1.


2.


3.


4.


5.


6.

7.

ᒥᓂᔅᑕ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ

ᐊᖏᖅᑕᐅᔪᑦ

ᐊᕕᒃᑐᖅᓯᒪᔪᓂ ᐆᒪᔪᓕᕆᔨᖕᑯᑦ ᑲᑎᒪᔩᖏᑦ

ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 2021-ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᖃᐅᔨᓴᕈᑎᕕᓃᑦ

ᑖᓐᓇ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐅᓂᒃᑳᖅ ᐱᔭᕇᖅᑕᐅᓚᐅᖅᑐᖅ ᐱᓕᕆᖃᑎᒌᒍᑕᐅᓪᓗᓂ ᐅᑯᓇᙵᑦ:

- ᔫᓯᐱ ᓅᐊᑦᑫᑉ, ᐋᓐᑎᐆᕆᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓴᑦ ᐋᒻᒪ ᓇᐸᖅᑐᓕᕆᔨ
- ᓂᑯᓚᔅ ᓚᓐ, ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓ ᑲᓇᑕᒥᑦ
- ᑭᐊᕕᓐ ᒥᑐᓪ, ᐋᓐᑎᐅᕆᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓴᑦ ᐋᒻᒪ ᓇᐸᖅᑐᓕᕆᔨ
- ᒫᑎᓐ ᐆᐹᑦ, ᐋᓐᑎᐅᕆᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓴᑦ ᐋᒻᒪ ᓇᐸᖅᑐᓕᕆᔨ
- ᑕᐃᓚ ᖀᔅ, York ᓯᓚᑦᑐᖅᓴᕐᕕᒡᔪᐊᖅ
- Guillaume Szor, Québec Ministère des Forêts
- ᓚᐃᐅᓪ ᐅᐊᓪᑕᓐ, ᐋᓐᑎᐅᕆᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓴᑦ ᐋᒻᒪ ᓇᐸᖅᑐᓕᕆᔨ
- ᔮᓯᒪ ᕕᐅᕐ, ᓄᓇᕗᑦ ᑲᕙᒪᖓ

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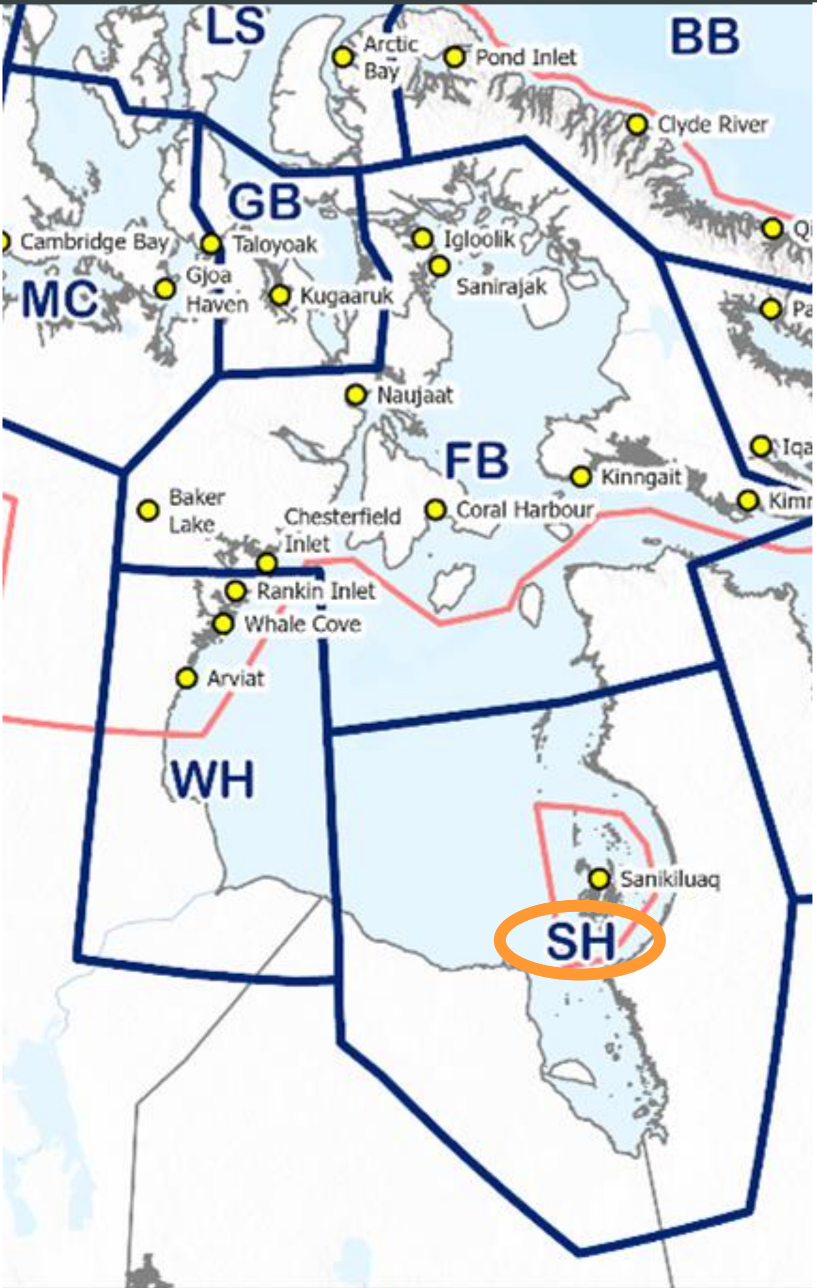
ᖁᔭᓐᓇᒦᒃ

ᐱᓕᕆᐊᖅ ᐱᐊᓂᒍᓐᓇᕋᔭᓚᐅᖕᒋᑦᑐᖅ ᐃᓚᐅᖅᑲᑕᐅᙱᑉᐸᑕ ᐊᒻᒪᓗ ᐃᑲᔪᖅᑕᐅᓚᐅᖕᒋᑉᐸᑕ ᐆᒪᔪᓕᕆᔨᐊᓖᒃᑯᑦ ᐊᒻᒪᓗ ᐅᑯᐊ ᑕᑯᓯᒪᔪᑦ:

- S. ᐅᐃᑖᓗᒃᑐᖅ
- Q. ᓇᐹᖅᑐᖅ
- J. ᑲᓴᓪᓗᐊᖅ
- M. ᐊᕐᓇᓱᒃ
- P. ᐃᑉᐸᒃ
- J. ᐃᑉᐸᒃ
- C. ᑕᑲᑕᒃ

ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᐃᑦ



- ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ 2011–ᒥ ᐊᒻᒪ 2016–

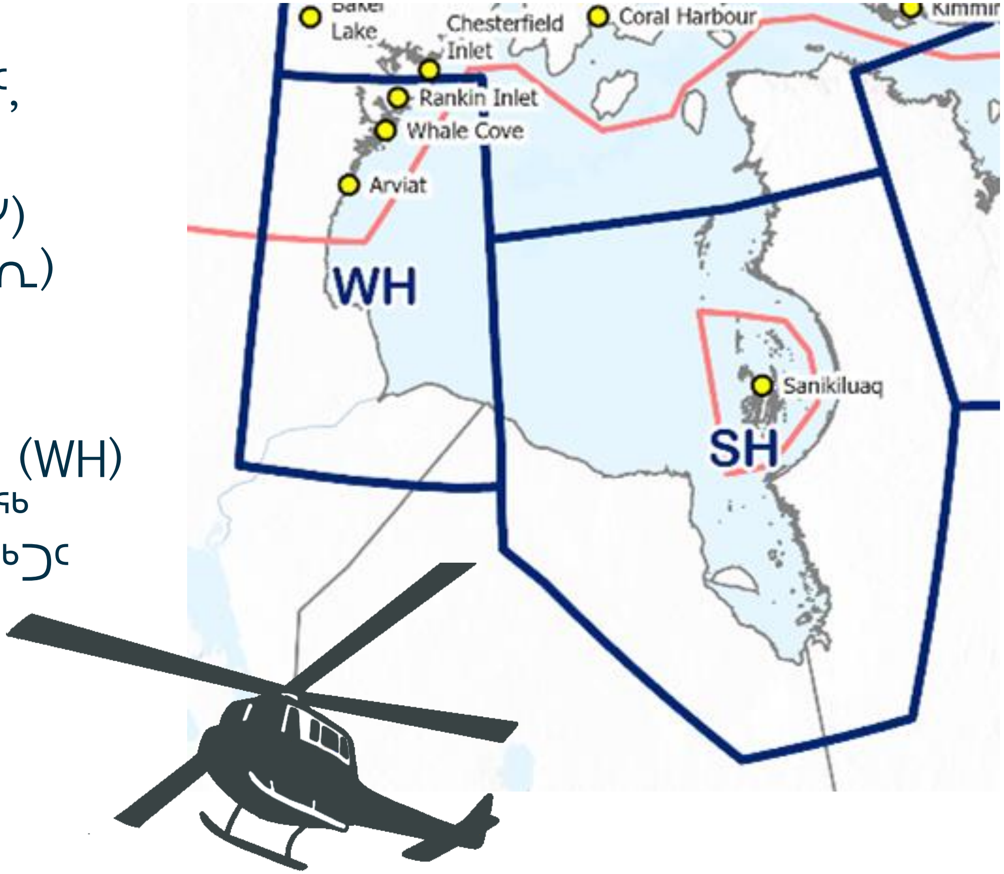
- ᐲᖅᓯᓂᖅ ᓇᐅᒃᑲᖅᑕᐅᑎᓄᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᓕᕆᐊᖑᓪᓗᓂ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᐊᒻᒪ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᐊᒻᒪ ᐋᓐᑎᐅᕆᐅᒥ ᓄᓇᒥᐅᑕᓕᕆᔨᒃᑯᓐᓂ ᐊᒻᒪ ᓇᐹᖅᑐᓕᕆᔨᒃᑯᓐᓂ
  - ᐱᖓᓐᓇᖅᐸᓯᐊᓂᒃ ᓂᒋᖅᐸᓯᐊᓂᓗ ᑕᓯᐅᔭᕐᔪᐊᒥ

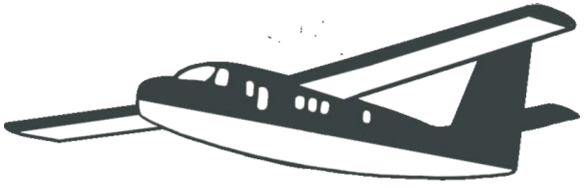
- ᐊᖏᖃᑎᒌᒍᑎ ᐅᓄᕐᓂᖏᑦ **ᐃᓄᐃᑦ–ᓇᓄᐃᑦ ᐊᐳᖃᑦᑕᐅᑎᕙᒃᑐᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᖏᓐᓇᕐᓂᖏᓐᓂᒃ ᖃᖓᓐᖑᖅᐸᓪᓕᐊᑎᓪᓗᒍ**

- ᖃᐅᔨᓴᖅᑕᐅᒐᔪᒡᓗᓂ ᖃᐅᔨᓂᐊᕐᓗᓂ **ᐊᓯᔾᔨᑳᓪᓚᖕᓂᐅᔪᓂᑦ** ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

- ᓈᓴᐃᓚᐅᖅᑐᑦ 2011-ᒥᑦ, 2016-ᒥᑦ, ᐊᒻᒪᓗ 2021-ᒥᑦ
- ᓈᓴᐃᓚᐅᖅᑐᑦ ᐊᑯᓪᓕᕈᕐᕕᒃ (ᐋᑭᓯ) ᓄᙳᐊᓂᑦ – ᐊᒥᕋᐃᔭᕐᕕᒃ (ᓯᑎᐱᓈ) ᐱᒋᐊᓵᕐᓂᖓᓄᑦ
  - ᓇᓄᐃᑦ ᓄᓇᒦᑦᑐᑦ
- ᐱᖓᖕᓇᖅᐸᓯᖓᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ (WH) ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᐊᓂᑦ ᑕᓯᐅᔭᕐᔪᐊᖅ (SH) ᐊᑕᐅᑦᑎᒃᑯᑦ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ



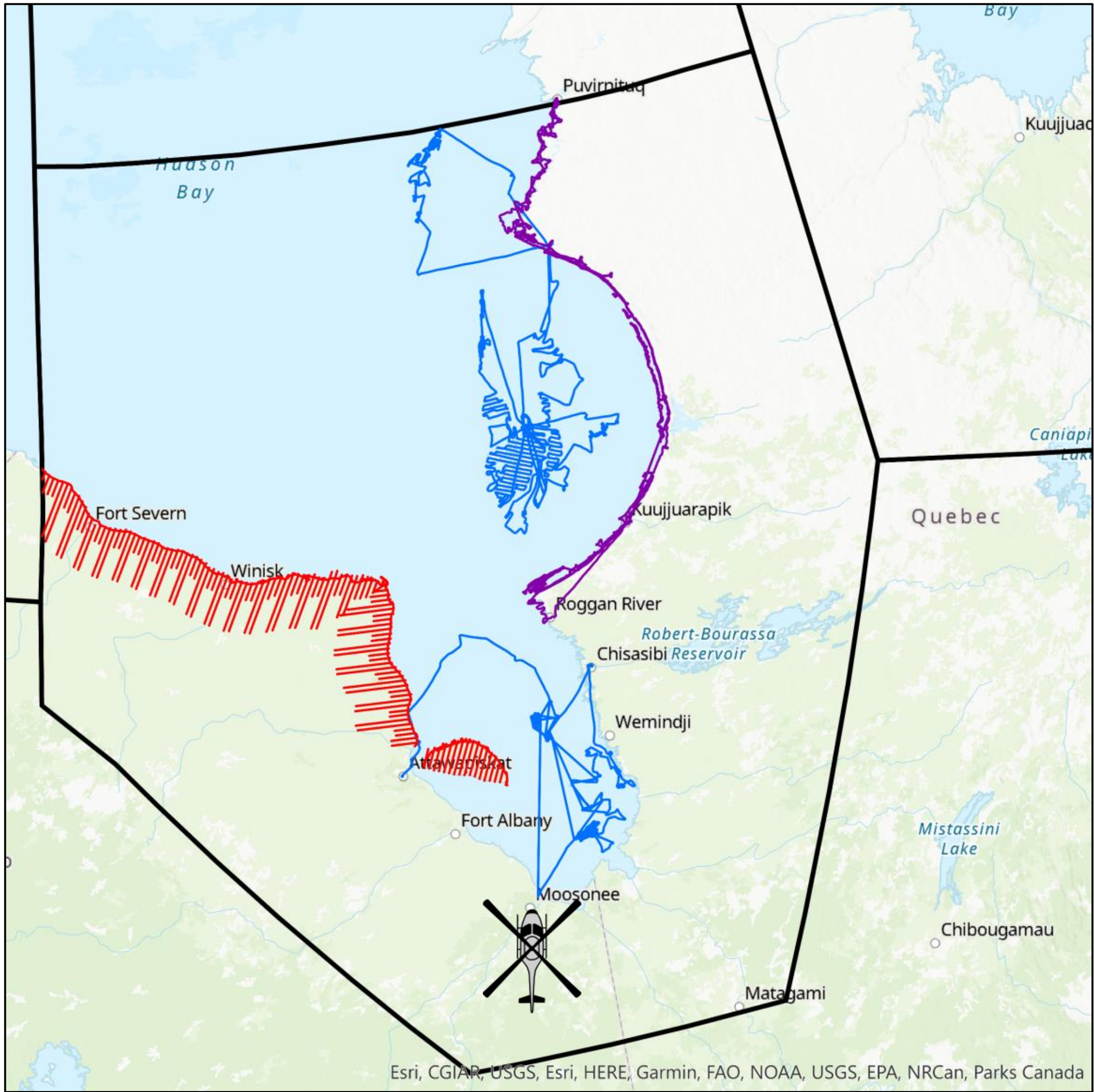


ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

- ᑲᑎᖅᓱᕐᓗᒋᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᑦ **ᒥᒃᓴᐅᓴᓐᓂᐊᕐᓗᒋᑦ ᐊᒥᓲᓂᖏᑦ** ᐆᒪᔪᑦ
- **ᐅᖓᓯᒋᔭᒥ ᖃᐅᔨᓴᕐᓂᖅ** ᐊᒻᒪ **ᒪᕐᕉᖕᓂᒃ ᖁᖖᒋᐊᖅᑎᖃᕐᓗᑎᒃ** ᖃᐅᔨᓵᖅᑐᑦ
- **ᐊᒡᒍᖅᓯᒪᓂᖏᑦ ᐊᒻᒪ ᑭᑉᐹᓇᒃᓯᓚᖅᓯᒪᔪᑦ** ᑐᙵᕕᒋᓪᓗᒋᑦ ᑭᖑᓂᖔᑦᑎᓐᓂ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᓚᐅᖅᑐᑦ, ᐊᖑᔭᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᑦ, ᐃᓚᓕᐅᑎᓂᐊᖅᐸᕗᓪᓗ ᓄᓇᓕᖕᓂ ᑭᐅᔾᔪᑕᐅᖃᑦᑕᓚᐅᖅᑐᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

- ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᑭᑉᐹᕆᒃᓯᓕᖅᓯᒪᔪᓂᒃ ᓱᓕ ᖃᐅᔨᓴᖅᑕᐅᒃᑲᓐᓂᖅᑐᑦ 2011-ᒥᑦ 2016-ᓗ
  - ᑭᓯᐊᓂ ᑯᐸᐃᖕᒥ ᓄᓇᖅᐸᓯᒃᑐᑦ ᑲᒪᒋᔭᐅᙱᓐᓂᖅᓴᐅᒐᓗᐊᖅᑎᓪᓗᒋᑦ ᓄᓇᓕᖕᓂ ᐅᖃᖃᑎᖃᓚᐅᖅᑎᓪᓗᒋᑦ
- ᑭᑉᐹᕆᒃᓯᓕᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᑦ ᐊᑐᖅᓗᑎᒃ:
  - 2011-ᒥ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᑦ ᖃᓄᐃᓕᖓᓂᖏᓪᓗ
  - ᐱᖅᑯᓯᕆᔭᐅᔪᒥᒃ ᖃᐅᔨᒪᓂᖅ
  - ᖃᐅᔨᒋᐊᖅᕕᒃᓴᐅᔪᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᑦ

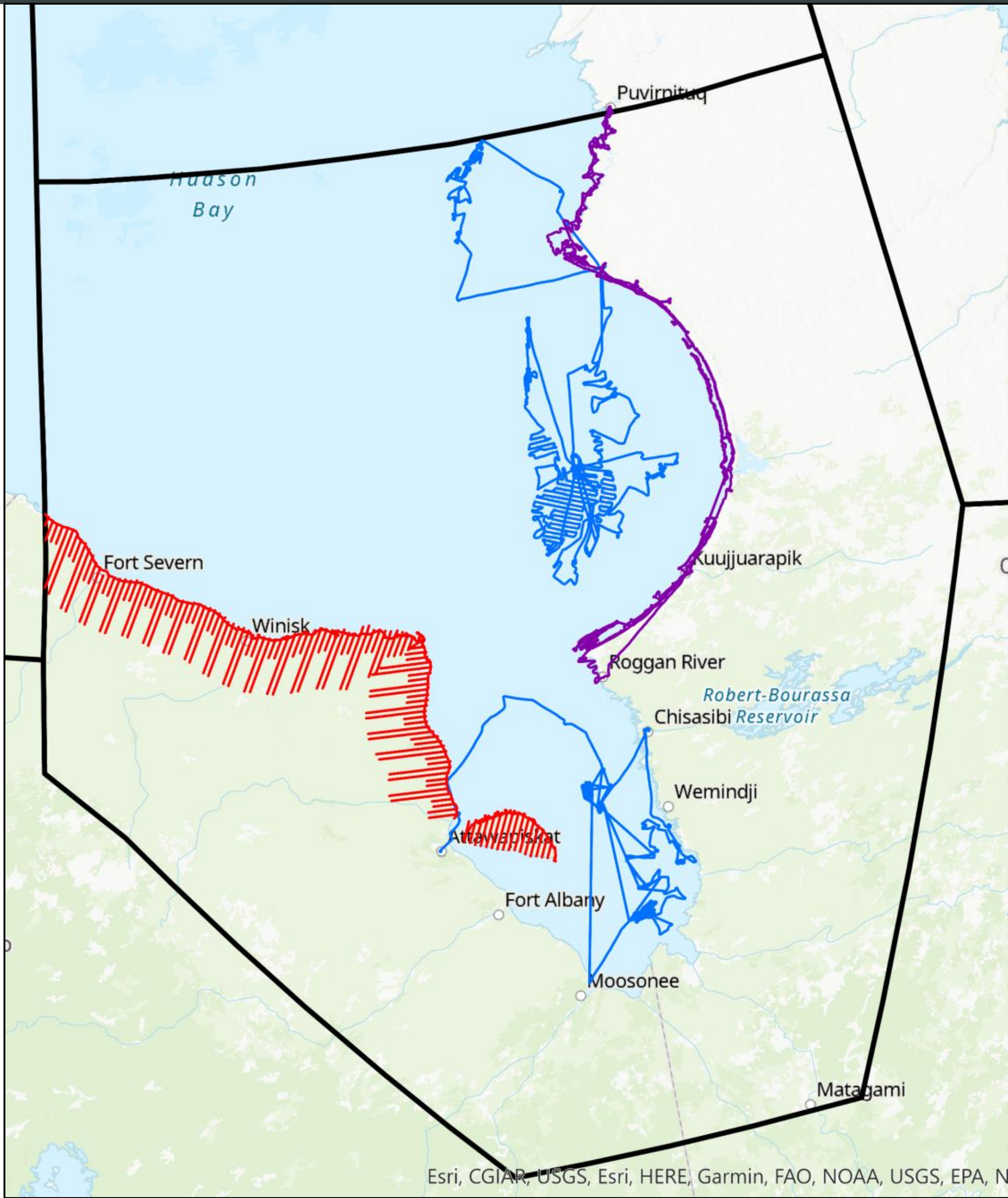


Esri, CGIAR, USGS, Esri, HERE, Garmin, FAO, NOAA, USGS, EPA, NRCan, Parks Canada

ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᐃᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

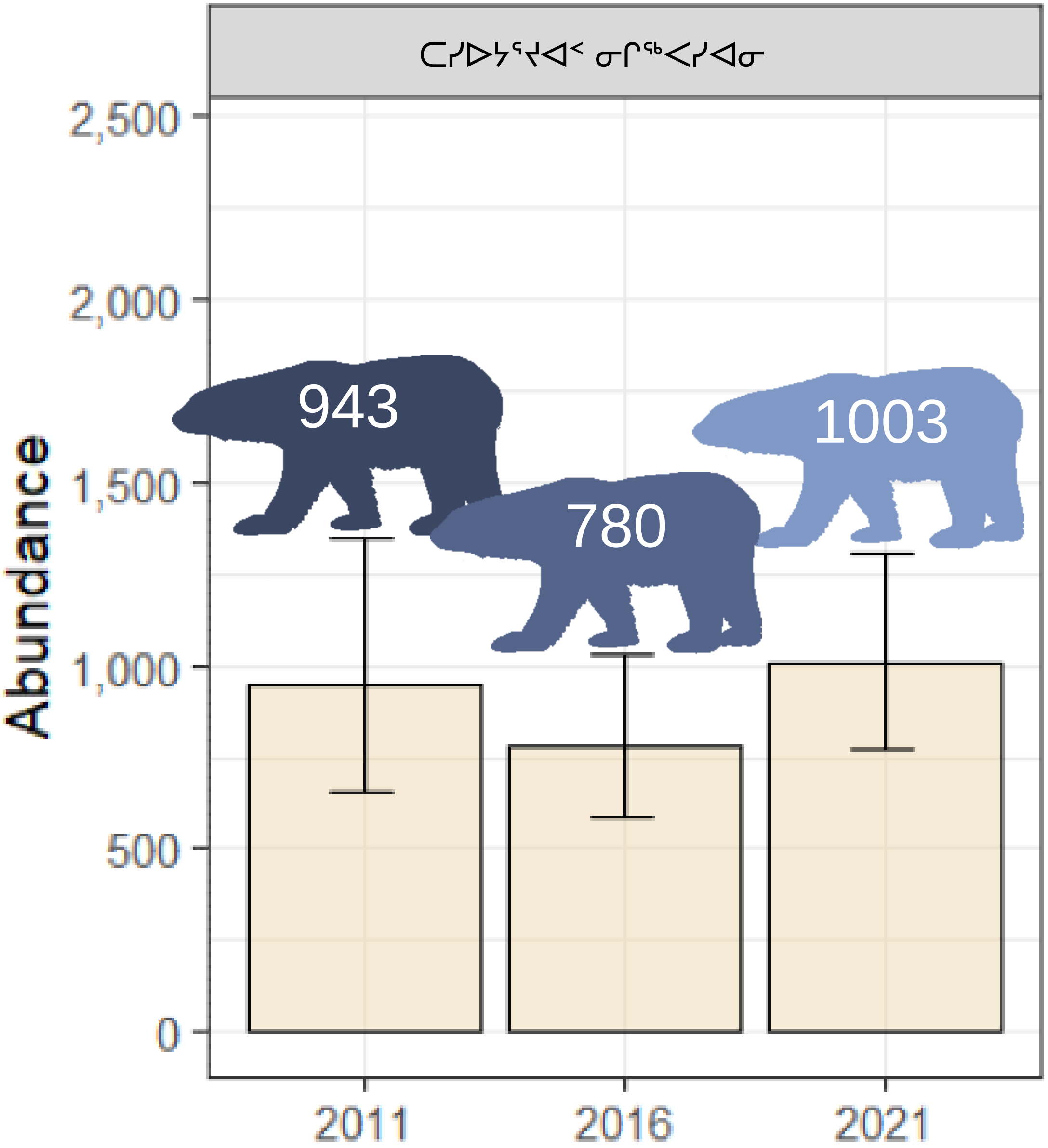
- ᑕᑯᖃᑦᑕᖅᓯᒪᔪᑦ ᐅᓄᕐᓂᖅᓴᓂᒃ ᓇᓄᕐᓂᒃ ᑲᑎᕐᖢᒋᑦ 2016−ᖑᓚᐅᖅᑎᓪᓗᒍ ᐊᔾᔨᐸᓗᒋᓪᓗᓂᒋᓪᓗ 2011−ᒥ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᓄᑦ
- 138-ᓂᒃ ᑲᑎᙵᔪᓂᒃ ᓇᓄᕐᓂᒃ ᑕᑯᔪᖃᓚᐅᖅᑐᖅ
- ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ 2016−ᒧᑦ: 1003 (773 – 1302)
- 29%−ᒥ ᐅᓄᖅᓯᒋᐊᖅᓯᒪᔪᑦ 2016-ᖑᓚᐅᖅᑎᓪᓗᒍ



ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᐃᑦ

ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

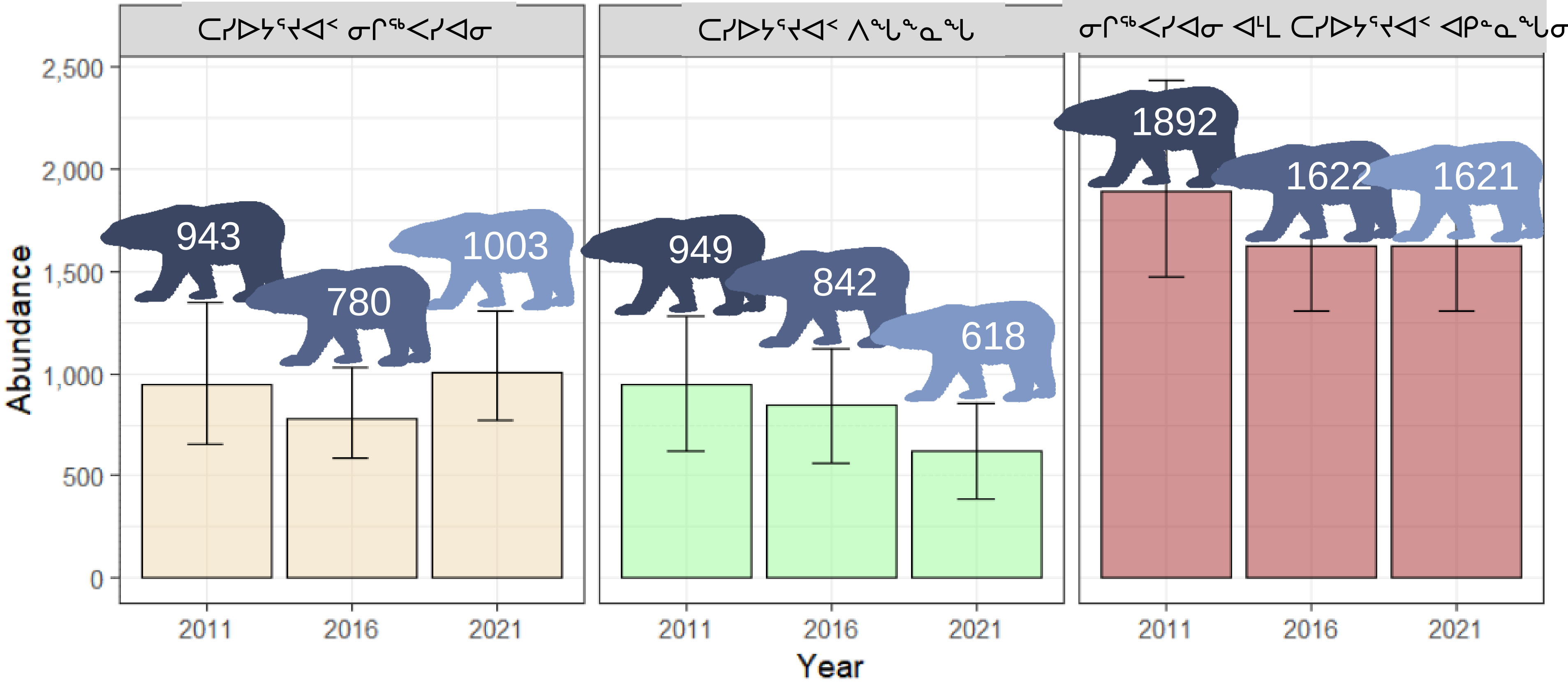
- ᑕᑯᖃᑦᑕᖅᓯᒪᕗᑦ ᐅᓄᕐᓂᖅᓴᓂᒃ ᓇᓄᕐᓂᒃ ᑲᑎᔾᖢᒋᑦ 2016−ᖑᓚᐅᖅᑎᓪᓗᒍ ᐊᔾᔨᐸᓗᒋᓪᓗᓂᒋᓪᓗ 2011−ᒥ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᓄᑦ
- 138-ᓂᒃ ᑲᑎᖖᒐᔪᓂᒃ ᓇᓄᕐᓂᒃ ᑕᑯᔪᖃᓚᐅᖅᑐᖅ
- ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ 2016−ᒧᑦ ᑎᑭᔾᖢᒍ: 1003 (773 – 1302)
- 29%−ᒥ ᐅᓄᖅᓯᒋᐊᖅᓯᒪᔪᑦ 2016-ᖑᓚᐅᖅᑎᓪᓗᒍ



ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᐃᑦ

ᖃᓐᓚᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᑦ

- ᐱᕝᕙᑕᐅᑐᐃᓐᓇᑎᐊᓖᑦ ᐅᓄᖅᓯᒋᐊᖅᑐᓄᑦ?
  - ᐊᕐᖀᒍᑕᒫᑦ ᓄᒃᑕᕐᓂᖏᓐᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᐊᓄᑦ?



ᐲᖅᓯᓂᖅ ᖃᐅᔨᓴᖅᑕᐅᔪᒃᓴᒥᒃ ᓯᓂᒃᑎᑕᐅᓗᑎᒃ

Environment and Climate Change Canada
Environnement et Changement climatique Canada

Distributional shifts of polar bears (*Ursus maritimus*) in Hudson Bay in relation to sea ice dynamics, 2017-2022 Final Report

D. McGeachy[1,4], N. J. Lunn[1], J. M. Northrup[2], V. Trim[3], C. Davis[4], A. E Derocher[4]

[1] Environment and Climate Change Canada, CW-405 Biological Sciences Building, University of Alberta, Edmonton, AB T6G 2E9
[2] Ontario Ministry of Natural Resources and Forestry, DNA Building B217, Trent University, 2140 East Bank Drive, Peterborough, ON K9L 1Z8
[3] Manitoba Department of Natural Resources and Northern Development, Water Stewardship and Biodiversity Division, Wildlife and Fisheries Branch, Box 28, 59 Elizabeth Drive, Thompson, MB R8N 1X4
[4] Department of Biological Sciences, University of Alberta, Edmonton, AB T6G 2E9

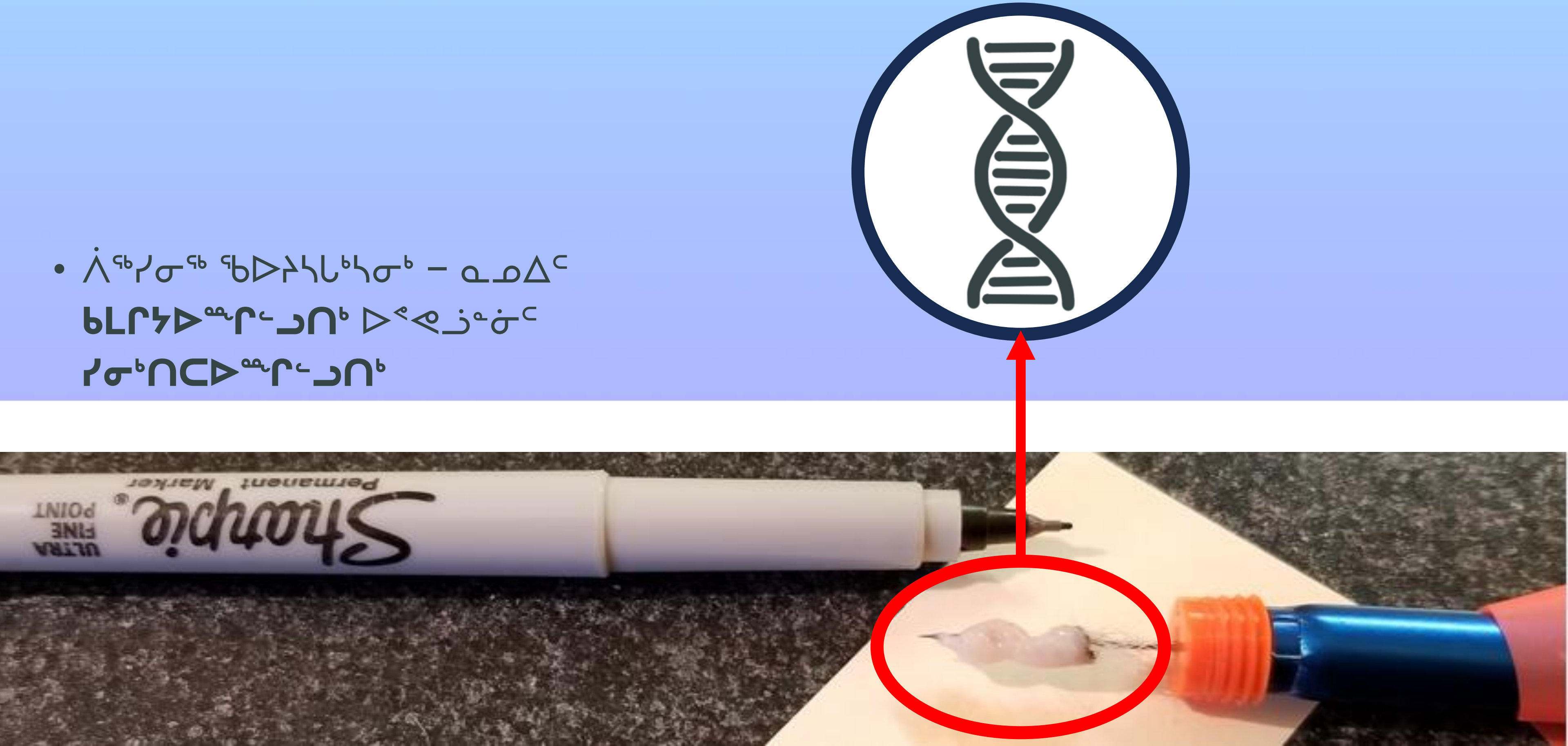
ᑕᐃᕕᑦ ᒪᒌᓯ
ᓇᓄᓕᕆᔨ
ᐊᕙᑎᓕᕆᓂᕐᒧᑦ ᐊᒻᒪ ᓯᓚᐅᑉ
ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥᑦ

ᖃᐅᔨᓴᒐᒃᓴᓂᒃ ᓯᓂᓕᖅᑎᑕᐅᖕᒋᔅᓗᓂ

ᖃᐅᔨᓴᒐᒃᓴᓂᒃ ᓯᓂᓚᖅᑎᑕᐅᙱᔾᔪᓂ

- ᐲᖅᓯᓂᖅ ᖃᐅᔨᓴᒐᒃᓴᓂᒃ – ᓇᓄᐃᑦ **ᑲᒪᒋᔭᐅᙱᑦᑐᑎᒃ** ᐅᕝᕙᓘᓐᓃᑦ **ᓯᓂᒃᑎᑕᐅᙱᑦᑐᑎᒃ**

ᓯᕗᓚᖅᓯᑭᑏᑦ/ᓇᑭᙵᕐᓂᖏᑦ

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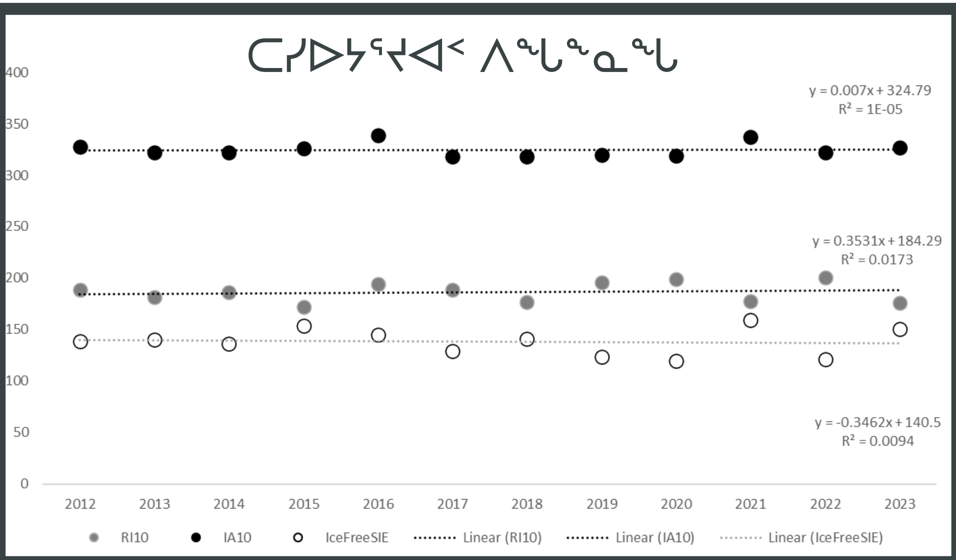
Hudson
Bay
WH
SH
DA

ᖃᓄᐃᓕᖅᓂᖏᑦ - ᑕᕆᐅᒥ ᓯᑯᑦ ᐃᓕᖓᓐᓂᕆᔭᖏᑦ



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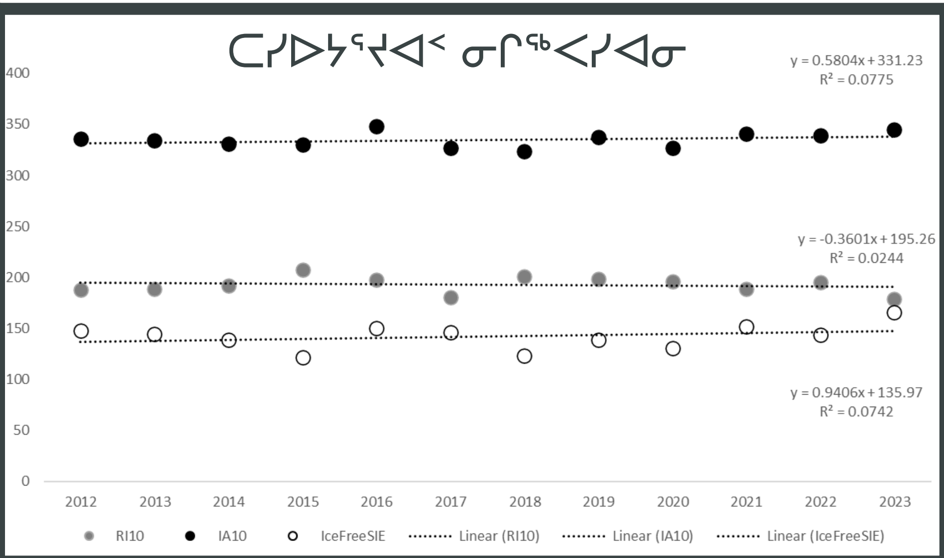

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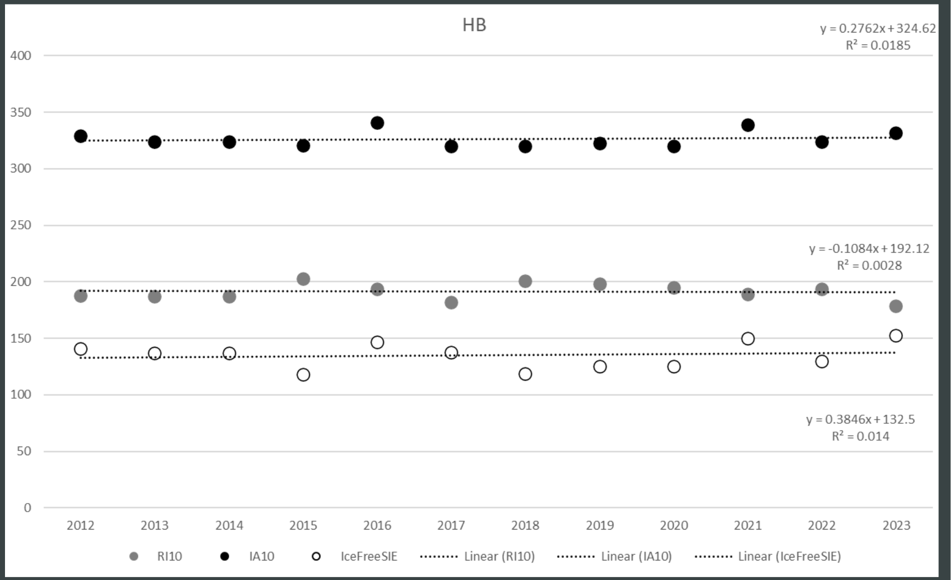
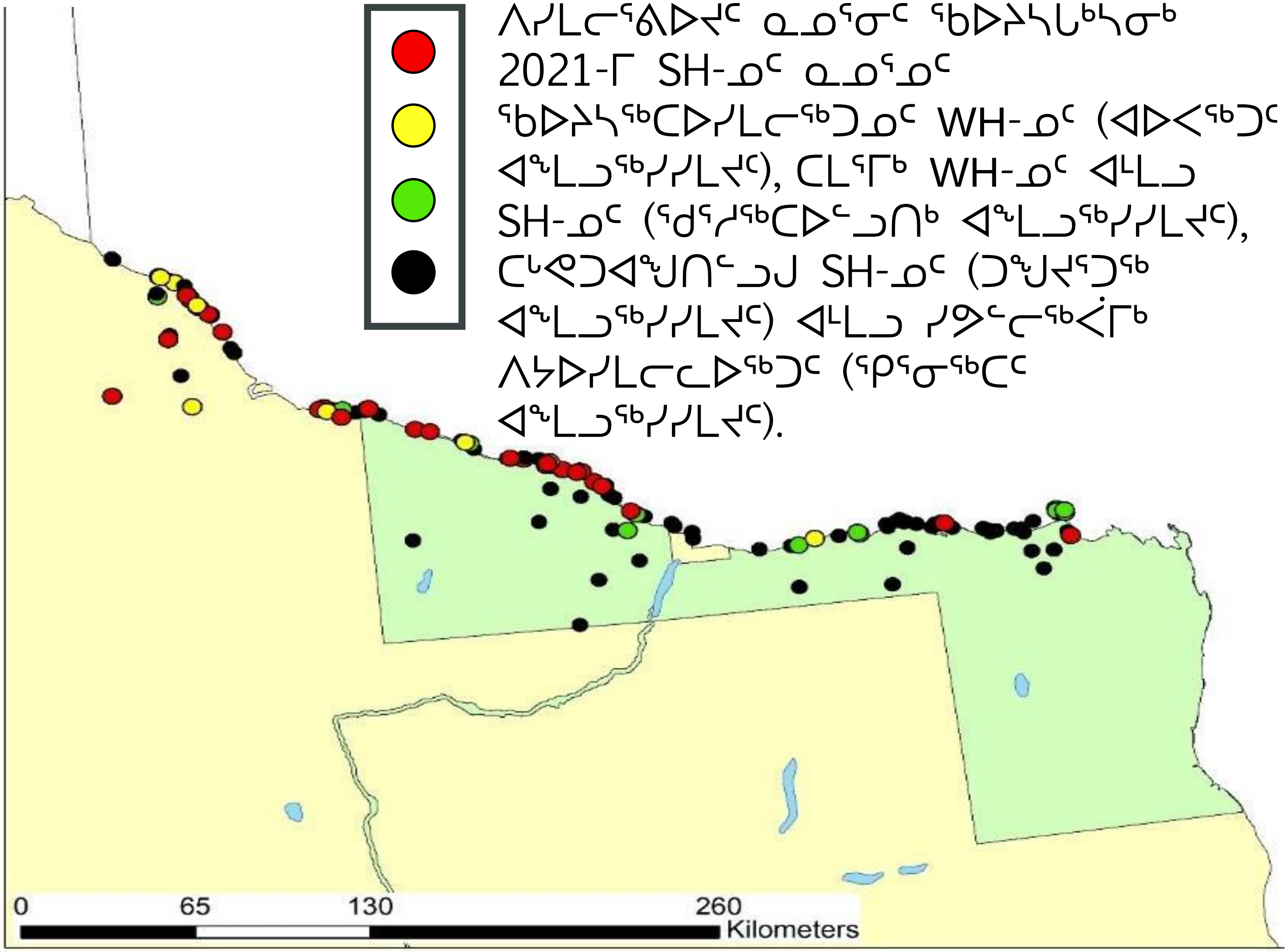


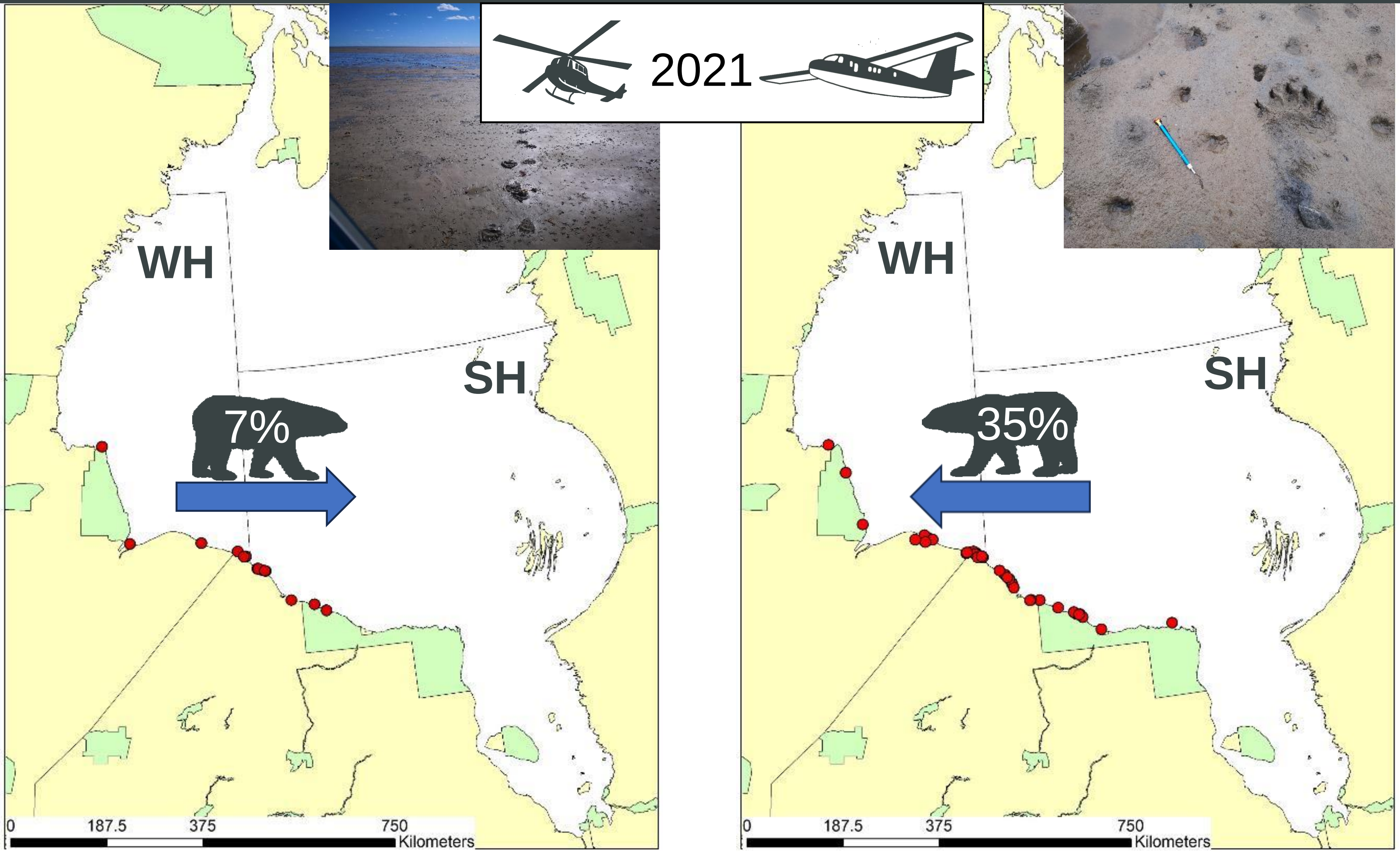
HB

ᖃᓄᐃᓕᕐᓂᖏᑦ - 2021 SH-ᓄᑦ ᒫᓐᓇᒧᑦ ᑎᑭᓪᓗᒍ

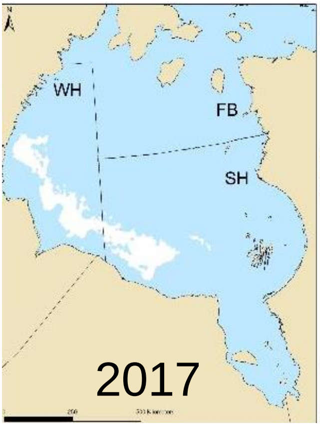


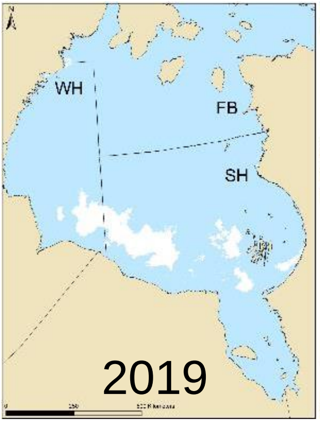
ᑎᑎᕋᐅᔭᖅᓯᒪᔪᖅ 6. ᖃᐅᔨᓴᑌᑉᓴᓂᒃ ᓇᑭᑦ ᐱᓯᒪᓕᕐᕕᐅᔪᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᑌᑉᓴᓂᒃ 2021-ᒥ SH-ᓄᑦ ᓇᓄᕐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᓕᖅᑐᓄᑦ WH-ᓄᑦ (ᐊᐅᐸᖅᑐᑦ ᐊᖓᓗᖅᓯᓯᒪᔪᑦ), ᑕᒪᕐᒥᒃ WH-ᓄᑦ ᐊᒻᒪᓗ SH-ᓄᑦ (ᖁᕐᓱᖅᑕᐅᓪᓗᑎᒃ ᐊᖓᓗᖅᓯᓯᒪᔪᑦ), ᑕᒡᕙᑐᐊᖑᑎᓪᓗᒍ SH-ᓄᑦ (ᑐᖑᔪᕐᑐᖅ ᐊᖓᓗᖅᓯᓯᒪᔪᑦ) ᐊᒻᒪᓗ ᓯᕗᓪᓕᖅᐹᒥᒃ ᐱᔭᐅᓯᒪᓕᓚᐅᖅᑐᑦ (ᕿᕐᓂᖅᑕᑦ ᐊᖓᓗᖅᓯᓯᒪᔪᑦ).

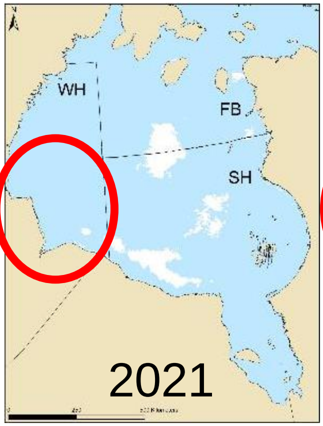
ᖃᓄᐃᓕᖅᓂᖏᑦ - WH-ᓂᑦ SH-ᒧᑦ
ᐊᒻᒪᓗ SH-ᓂᑦ WH-ᓂᑦ 2021/2022

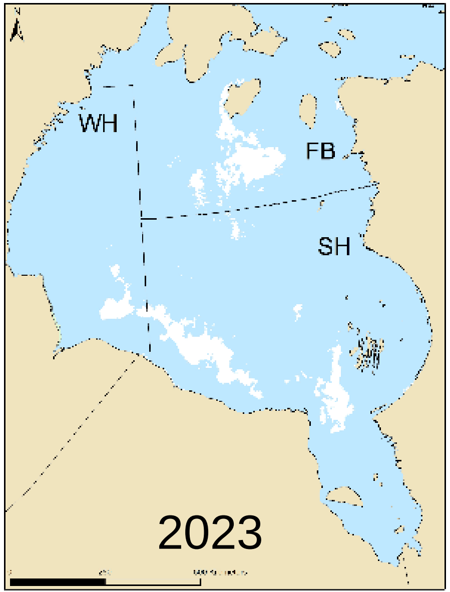


ᖃᓄᐃᓕᕐᓂᖕᒥᑦ - ᑮᖕᒍᓪᓕᖅᐸᐅᔪᖅ ᐊᕙᑎᒋᔭᐅᔪᒥᑦ ᑲᑎᓘᒃᑖᕐᓗᒥᑦ ᓯᑯᖃᕐᓂᕐᒥᑦ ᐊᐅᔭᓚᐅᖅᑳᖅᑎᖕᓇᒍ

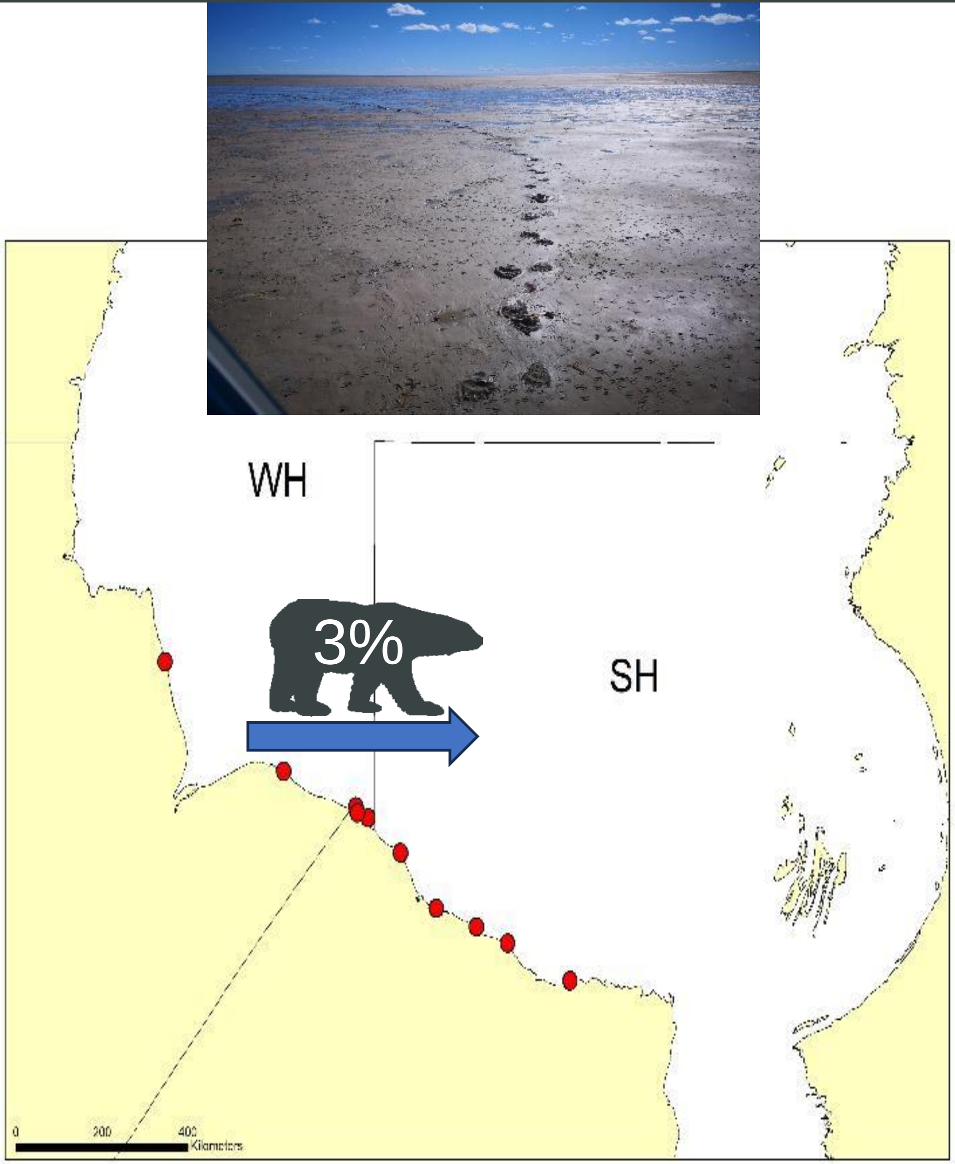


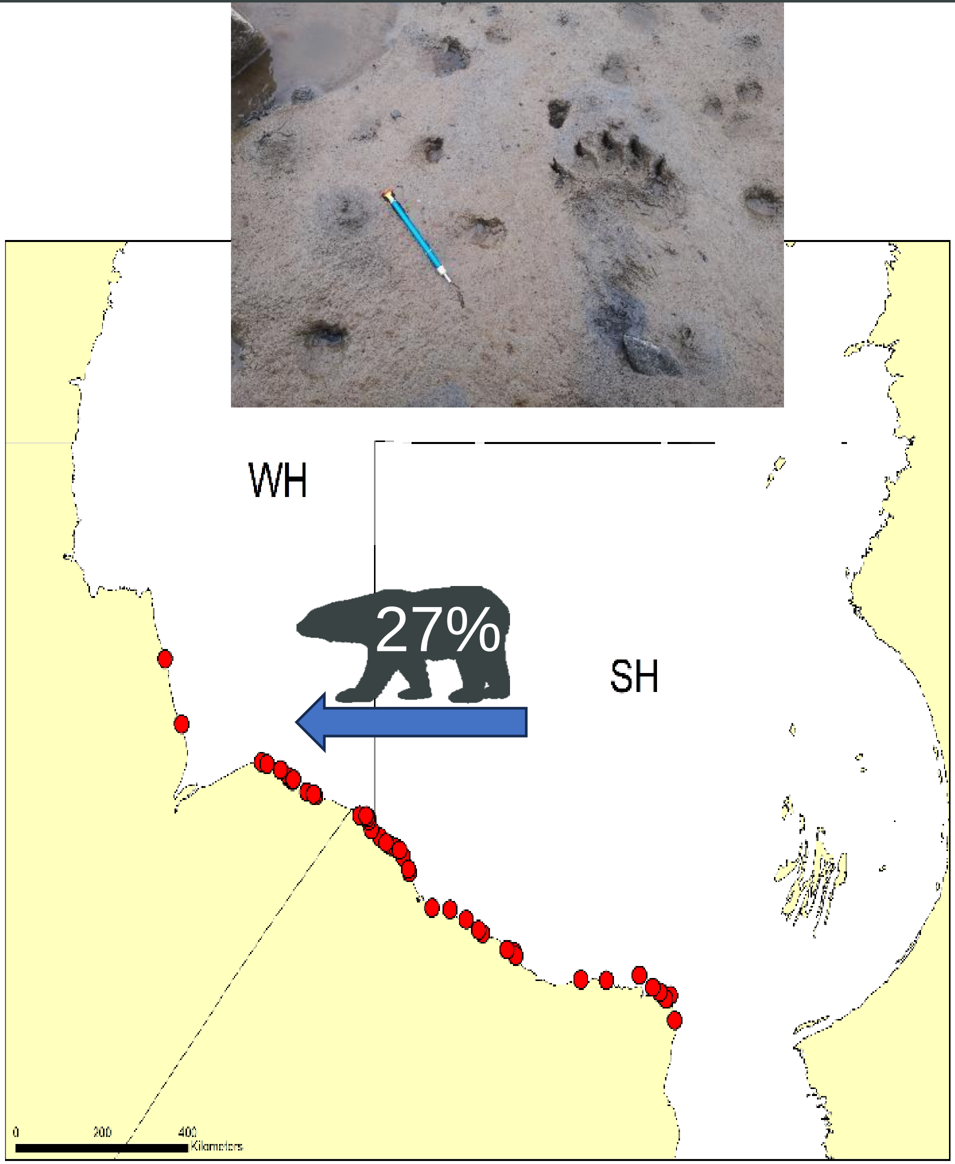




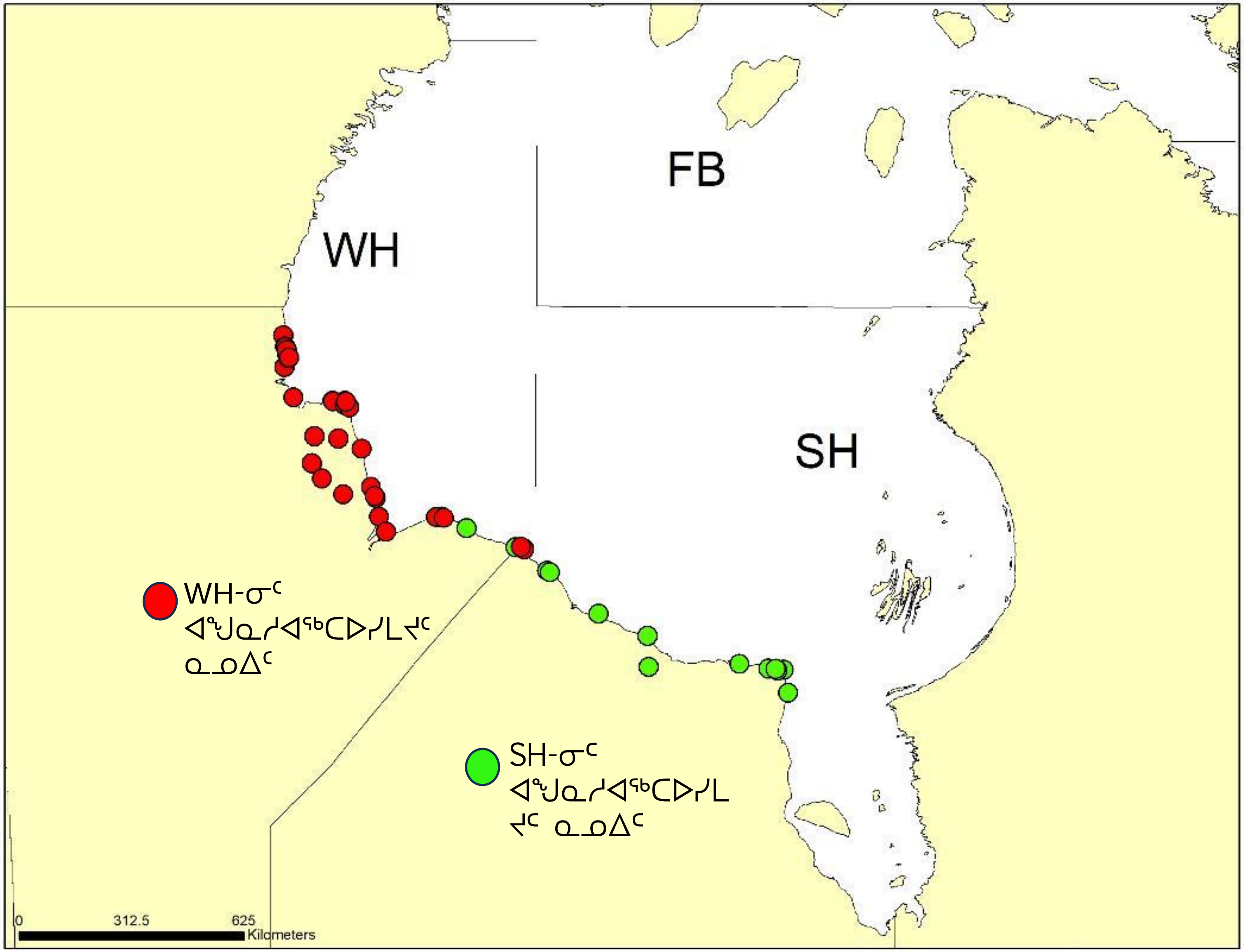


ᖃᓄᐃᓕᖅᓂᖕᒋᑦ - WH-ᓂᑦ SH-ᓄᑦ ᐊᒻᒪᓗ SH-ᓂᑦ WH-ᓂᑦ 2022/2023





WH-ᓂᑦ ᐊᒻᒪᓗ SH-ᒧᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᑎᒃᓴᓂᒃ ᐱᓯᒪᓕᓚᐅᖅᑐᑦ 2012-2023



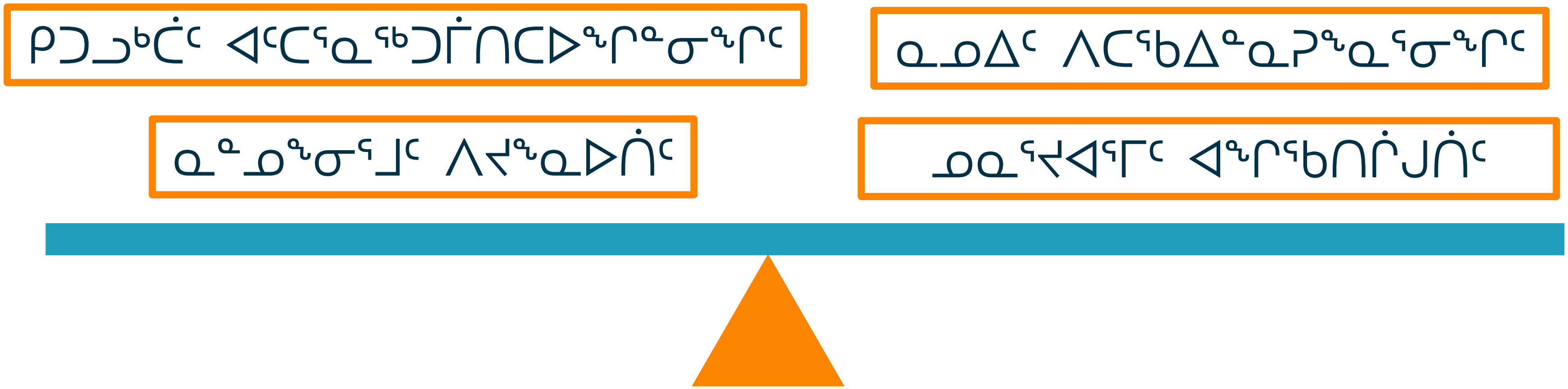
🟥 WH-ᓂᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ

🟩 SH-ᓂᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ

ᐱᓕᕆᐊᖑᓂᐊᓕᕐᒥᔪᑦ – ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ

ᐱᓕᕆᐊᖓᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ

- ᐃᒡᓗᒌᖓᓕᕆᓂᖅ



ᑭᑐᓗᒃᑖᑦ ᐊᑦᑕᕐᓇᖅᑐᒦᑎᑕᐅᖏᓐᓂᖏᑦ

ᓇᓄᐃᑦ ᐱᑕᖃᐃᓐᓇᕈᖕᓇᕐᓂᖏᑦ

ᓇᓐᓄᖕᓂᕐᒧᑦ ᐱᔪᖕᓇᐅᑏᑦ

ᓄᓇᕐᔪᐊᕐᒥᑦ ᐊᖏᕐᑲᑎᕌᒍᑏᑦ

ᑲᑎᓗᒃᑖᖅᓱᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᖕᓇᖅᑐᑦ (TAH)

- ᒫᓐᓇ ᑲᑎᔅᓱᒋᑦ ᖃᔅᓯᕌᕈᓐᓇᖅᑎᑕᐅᓂᖅ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 25−ᖑᕗᑦ ᓇᓄᐃᑦ ᐊᕐᕌᒍᒥ ᐊᑕᐅᓯᕐᒥ.
- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᒫᓐᓇ ᐊᑐᕐᑯᔨᖕᒪᑕ **ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ 25−ᓂᑦ.**
  - ᑕᒪᓐᓇ ᑕᐃᒪᐃᔾᔪᑎᖃᖅᐳᖅ ECCC−ᒧᑦ ᑎᒥᖏᓐᓂᑦ ᐹᖅᓯᓂᕐᒧᑦ ᖃᓄᐃᓕᖕᒪᓕᖅᑎᑕᐅᔪᓄᑦ **ᑕᑯᒃᓴᐅᑎᑦᑎᓪᓗᓂ ᓇᒧᙵᐅᕙᓪᓕᐊᓂᕆᔭᐅᔪᒥᒃ ᐅᑯᓇᙵᑦ, ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖅᐸᓯᐊᓂ ᐊᒻᒪ ᓂᒋᖅᐸᓯᐊᓂ**

ᐱᓕᕆᐊᖑᓂᐊᓕᖅᒥᔪᑦ – ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ

- ᐊᖕᒋᖃᑕᐅᕕᑦ ᐊᒥᓲᓂᖕᒋᑦ ᓇᓄᐃᑦ **ᐊᑕᐅᓯᑦᑕᐃᓐᓇᓕᓂᖕᒋᓐᓂᒃ** ᖃᑯᒍᖖᒍᖅᐸᓪᓕᐊᑎᓪᓗᒍ?
- **ᓈᒻᒪᒃᐸᑦ** ᓇᓐᓄᒃᑕᐅᔪᒃᓴᑦ? **ᐊᒥᓲᖖᒋᓗᐊᕐᐸᑦ? ᐊᒥᓲᓗᐊᖅᐸᑦ?**
- ᑭᓱᒥᓪᓗ ᐅᔾᔩᕆᓯᓚᐅᖅᐱᑦ ᓇᓅᑉ **ᑎᒥᐊᑕ ᖃᓄᐃᓐᓂᖕᒐᓂ** ᖃᑯᒍᖖᒍᖅᐸᓪᓕᐊᑎᓪᓗᒍ?
- ᐊᔾᔩᐅᖖᒋᑦᑐᒥᑦ ᐅᔾᔩᕆᓯᓚᐅᖅᐱᑦ ᑐᓴᖅᑎᑦᑐᒪᓪᓗᑕᓗ?
- ᓇᐅᒃᑯᓪᓗ **ᐊᖕᒋᖃᑎᖃᕐᐱᑦ/ᐊᖕᒋᖃᑎᖃᖖᒋᓚᑎᑦ** ᖃᐅᔨᓴᑦᑎᓐᓂᑦ?

ᖁᔭᓐᓇᒦᒃ!

ᖃᐅᔨᒋᐊᕐᕕᒃᓴᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᑦ:

ᐊᓖᓴ ᐴᕼᐋᑦ

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ᔫ ᓄᐊᑦᕋᑉ

✉ joseph.northrup@ontario.ca

ᐊᐱᖅᑯᑎᒃᓴᖅᑲᖅᐲᑦ?

2021 Southern Hudson Bay polar bear subpopulation aerial survey – Final report

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[2]Environmental and Life Sciences Graduate Program, Trent University
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[5] Ministère de l'Environnement, de la Lutte contre les Changements Climatiques, de la Faune et des Parcs, Gouvernement du Québec
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[7]Department of Environment, Government of Nunavut

EXECUTIVE SUMMARY

The Southern Hudson Bay (SH) polar bear subpopulation is the most southern continuously occupied area for the species, covering the eastern and southern parts of Hudson Bay and all of James Bay, as well as corresponding parts of the Ontario and western Quebec coastline, and up to 120 km inland. Polar bears in this area are hunted by Indigenous peoples for food, cultural practices, and sale of hides. As such, assessing the abundance of polar bears in the SH subpopulation is critical for monitoring the impacts of climate change on polar bears, which is the main threat to polar bear persistence, and for ensuring sustainable harvest.

Early population inventories of the SH subpopulation used physical mark-recapture methods with surveys conducted approximately every ten years and concentrating mostly along the Ontario coastline. Results of these studies suggested that the population abundance along the Ontario coastline of Hudson Bay and some inland areas was largely similar from the 1980s to the mid-2000s, despite evidence of declines in body condition and survival related to declining sea ice. In 2011, partners agreed to a less invasive, but more frequent, monitoring approach based on aerial surveys. This approach was chosen due to concerns raised by Indigenous partners about handling polar bears as well as logistical and financial challenges of physical mark-recapture studies. As a result, starting in 2011, aerial surveys have been conducted every five years. The surveys combined distance sampling and double-observer mark-recapture aerial surveys of the coast, areas up to 60 km inland and offshore islands in Hudson and James Bays. The design and methodology were established according to the known distribution of SH bears during the ice-free season and the different survey approaches were used to sample across differences in bear density. Field work for the first survey was completed in Ontario in 2011 and Quebec in 2012. This survey methodology was then repeated in 2016 in both Ontario and Quebec. In keeping with management authority goals, a comprehensive aerial survey of SH was conducted in summer 2021 that maintained a nearly identical design as the previous surveys. This report presents the results of this third survey to provide a direct comparison across the three survey periods (2011-12, 2016 and 2021).

The 2021 aerial survey was flown from August 22 – September 1, using the same methods and design as the two previous surveys. The 2021 aerial survey produced two separate estimates: 1) 1003 (95% CI 773-1302), which is directly comparable to the previous 2016 survey abundance estimate and 2) 1119 (95% CI 860-1454), which provided a more robust estimate using recent advances in statistical methods. Both estimates indicated a greater number of bears within the SH subpopulation in 2021 than in 2016.

Two main biological drivers are likely to have contributed to the greater number of bears observed within SH in 2021 relative to 2016: annual variation in the on-land distribution of bears in SH and the neighbouring Western Hudson Bay (WH) subpopulation, and improved demographic rates. It is likely that both drivers have contributed to the observed differences in abundance, to varying degrees, although there is no definitive evidence to support either driver. There appears to have been some movement of bears into SH from the WH subpopulation in 2021 around the time of the survey. A minor variation in the distribution of bears around the SH and WH border could have influenced the SH abundance estimate and may not represent an actual growth of the subpopulation. It is also possible that the greater number of bears observed in 2021 compared to 2016 was influenced in part by reduced mortality, increased birth rate, or some combination of the two. The three years preceding the 2021 survey had the longest duration of sea ice in Hudson Bay of the last decade, and harvest was lower between 2016 and 2021 than between 2010 and 2015; both factors could have contributed potential growth of the SH subpopulation.

Monitoring polar bear populations in the face of ongoing climate warming is critical for providing local communities that rely on polar bears with additional information for harvest management decision-making. Continued monitoring of reproduction, survival, and inter-annual movements within and between the WH and SH subpopulations will be critical to continue to inform management during the intervals between aerial surveys.

RÉSUMÉ

La sous-population d’ours blancs du sud de la baie d’Hudson (SBH) se trouve dans la zone la plus méridionale continuellement occupée par l’espèce. Cette zone couvre les parties est et sud de la baie d’Hudson et toute la baie James, ainsi que les parties correspondantes des côtes de l’Ontario et de l’ouest du Québec, et jusqu’à 120 km à l’intérieur des terres. Dans cette région, les Autochtones chassent l’ours blanc à des fins d’alimentation, de pratiques culturelles et de la vente de peaux. L’évaluation de l’abondance de la sous-population du SBH est donc essentielle pour suivre les répercussions des changements climatiques sur l’espèce, soit la principale menace pour la persistance de l’ours blanc, et garantir une exploitation durable.

Les premiers inventaires de la sous-population du SBH utilisaient des méthodes de marquage-recapture physique, et des relevés étaient réalisés environ tous les dix ans, principalement le long de la côte de l’Ontario. Selon les résultats de ces études, l’abondance de la sous-population le long de la côte ontarienne de la baie d’Hudson et dans certaines zones intérieures était largement similaire des années 1980 au milieu des années 2000, malgré des signes de déclin de la condition physique et de la survie liés à la réduction de la glace de mer. En 2011, des partenaires ont accepté d’utiliser une approche de suivi moins invasive, mais plus fréquente, basée sur des inventaires aériens. Cette approche a été choisie à cause des préoccupations soulevées par les partenaires autochtones concernant la manipulation des ours blancs, de même que des défis logistiques et financiers liés aux études de marquage-recapture physique. Par conséquent, depuis 2011, des inventaires aériens sont réalisés tous les cinq ans. Les inventaires combinent la méthode d’échantillonnage par distance (« distance sampling ») et la méthode de marquage-recapture à double observateur et sont réalisés le long des zones côtières, dans certains secteurs jusqu’à 60 km à l’intérieur des terres et sur les îles au large de la baie d’Hudson et de la baie James. Le plan expérimental ainsi que la méthodologie d’inventaire ont été établies en fonction de la répartition connue des ours du SBH pendant la saison libre de glace, et les différentes approches d’échantillonnage ont été utilisées en fonction de la densité d’ours attendue dans les différents secteurs inventoriés. Le premier inventaire couvrant la totalité de la sous-population du SBH, a été réalisé sur deux années, couvrant l’Ontario en 2011 et le Québec en 2012. Un inventaire réalisé selon la même méthodologie a ensuite été répété en 2016 dans les deux provinces. Conformément aux objectifs des autorités de gestion, un inventaire aérien exhaustif de la sous-population a été mené à l’été 2021, selon une méthodologie presque identique à celle des inventaires précédents. Le présent rapport fait état des résultats de ce troisième inventaire afin de fournir une comparaison directe entre les trois périodes d’inventaire (2011-2012, 2016 et 2021).

L’inventaire aérien de 2021 a été réalisé du 22 août au 1er septembre à l’aide des mêmes méthodes et de la même conception que les deux inventaires précédents. Cet inventaire

a permis d’obtenir deux estimations distinctes : 1) 1 003 individus (IC à 95 % : 773-1 302), une estimation qui est directement comparable à l’estimation de l’abondance issue de l’inventaire précédent de 2016 et 2) 1 119 individus (IC à 95 % : 860-1 454), une estimation plus robuste obtenue selon une approche statistique novatrice. Les deux estimations indiquent un plus grand nombre d’ours au sein de la sous-population du SBH en 2021 qu’en 2016.

Deux facteurs biologiques principaux sont susceptibles d’avoir contribué à l’augmentation du nombre d’ours observés dans cette sous-population en 2021 par rapport à 2016 : une variation annuelle au niveau de la répartition terrestre des ours du SBH et de la sous-population voisine de l’ouest de la baie d’Hudson (OBH), et une amélioration des taux démographiques. Il est probable que ces deux facteurs aient contribué à des degrés variables aux différences observées de l’abondance, bien qu'il n'y ait pas de preuves définitives à l'appui de l'une ou l'autre de ces facteurs. Il semble y avoir eu un certain déplacement des ours de la sous-population de l’OBH vers celle du SBH en 2021 au moment de l’inventaire. Une variation mineure au niveau de la répartition des individus de part et d’autre de la frontière entre ces deux sous-populations pourrait avoir influé sur l’estimation de l’abondance de la sous-population du SBH et pourrait ne pas représenter une croissance réelle de celle-ci. Il est également possible que le plus grand nombre d’ours observés en 2021 comparativement à 2016 soit attribuable en partie à une réduction de la mortalité, à une augmentation du taux de natalité ou à une combinaison des deux. Les trois années précédant l’inventaire de 2021 ont connu la plus longue durée de la glace de mer dans la baie d’Hudson de la dernière décennie, et la récolte a été plus faible de 2016 à 2021 que de 2010 à 2015; ces deux facteurs pourraient avoir contribué à une croissance potentielle de la sous-population du SBH.

Le suivi des populations d’ours blancs dans le contexte du réchauffement climatique actuel est essentiel pour fournir aux communautés locales qui dépendent des ours blancs de l’information supplémentaire pour la prise de décision en matière de gestion des récoltes. La poursuite du suivi de la reproduction, de la survie et des déplacements interannuels au sein des sous-populations de l’OBH et du SBH, et entre elles, sera essentielle afin de fournir les informations nécessaires à la gestion de ces populations pendant les intervalles entre les inventaires aériens.

2021-ᒥ ᓯᕿᓂᕐᒥᐅᑦ ᑕᓯᐅᔭᕐᔪᐊᒥ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦᑕ ᖃᖓᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᑕᐅᓂᖓ

ᓀᓪᓕᑎᕐᓯᒪᔪᑦ

ᓯᕿᓂᕐᒥᐅᖏᑦ ᑕᓯᐅᔭᕐᔪᐊᒥ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦ ᓯᕿᓂᕐᒦᓂᕐᐸᐅᕗᑦ ᓇᔪᕐᑕᐅᖏᓐᓇᓕᒫᕐᓯᑎᒃ ᓇᓄᕐᓄᑦ, ᓯᕿᓂᐅᑉ ᓄᐃᕕᐊᓂ ᓯᕿᓂᖓᓂᐅᓂᕐᓴᓗ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐊᒻᒪᓗ ᔦᒥᓯ ᐯᓕᒫᑉ, ᐋᓐᑎᐊᕆᐆᒥᓪᓗ ᐃᓚᖓᓐᓂ ᐊᒻᒪᓗ ᑯᐯᒻᒥ ᓂᕿᓂᐅᑉ ᓂᐱᕕᐊᓂᐅᓂᕐᓴᖅ ᓯᓈᖏᓐᓂ, ᓄᓇᒧᑦ 120-ᑭᓚᒦᑕᓄᑦ ᑎᑭᑦᓯᓂ. ᓇᓄᐃᑦ ᑕᒪᒃᑯᓇᓂ ᓄᓇᓐᓂ ᐆᒪᔪᕐᓂᐅᑕᐅᓲᖑᕗᑦ ᓄᓇᖃᕐᖄᓯᒪᔪᓄᑦ ᓂᕿᑦᓴᐅᑎᓪᓗᒋᑦ, ᐃᓗᕐᖁᓯᒃᑯᑦ ᐱᕐᖁᓯᕐᒥᒍᑦ ᓄᓇᖃᕐᓂᓗ ᓂᐅᕕᐊᑦᓴᖃᕐᓂᓄᑦ. ᑌᒣᑦᑎᓗᒋᑦ, ᖃᐅᔨᓴᕐᓂᖅ ᓄᓇᖃᒃᓂᒥᒃ ᑕᓯᐅᔭᕐᔪᐊᒦᑦᑐᓂᒃ ᐱᓪᓚᕆᐅᔭᖅ ᐃᑉᐱᒋᔭᐅᓂᖏᓐᓄᑦ ᓱᕐᕋᑕᐅᓯᒪᓂᖏᑦ ᓯᓚᐅᑉ ᐊᓯᑦᔨᐸᓪᓕᐊᓂᖓᓄᑦ, ᑲᑉᐱᐊᓇᕐᑐᒦᑦᑎᓯᓂᕐᐸᐅᓯᓂ ᐱᑕᙯᓐᓇᓂᖏᓐᓂᒃ, ᓱᕐᖁᐃᓯᒪᓂᕐᒧᓗ ᐱᓪᓗᐊᑌᓕᒋᐊᖃᕐᓂᒧᑦ ᒪᙯᓐᓂᑯᑦ.

ᑭᓯᑕᐅᒪᐅᑏᑦ ᓯᕗᓪᓕᒥᓂᑌᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓯᕿᓂᖓᓂ ᓇᓄᐃᑦ ᓂᑦᑕᓚᕆᑦᑐᓂᒃ ᐊᑐᕐᓯᒪᔪᑦ ᖃᐅᔨᓴᐅᔭᒃᑯᑦ ᐊᑑᑎᔭᐅᔪᕕᓂᕐᓂᒃ ᐊᕐᕌᒍᒐᓛᑦ 10 ᐊᓂᒍᕐᑐᑦ ᐃᑉᐱᒋᔭᖃᓗᐊᖕᖑᐊᓱᑎᒃ ᐋᓐᑎᐊᕆᐅᑉ ᓯᓈᖏᓐᓃᑐᓂᒃ. ᓄᐃᑕᕕᓂᖏᑦ ᑕᒪᒃᑯᓇᓂ ᓇᓄᖃᕐᓂᖏᑦ ᐋᓐᑎᐊᕆᐆᑉ ᓯᓈᖏᓐᓂ ᑕᓯᐅᔭᕐᔪᐊᒥ ᓄᓇᑐᐃᓐᓇᐅᓂᕐᓴᖓᓂᓗ ᑌᒣᑦᑐᓴᓐᓇᒐᓛᕕᓃᑦ 1980 ᐊᕐᕌᒍᖏᓐᓂᓂᑦ 2000 ᐊᕐᕌᒍᖏᓐᓄᑦ ᑎᑭᑦᓯᒍ, ᑐᕚᖃᖕᖏᓂᕐᓴᐅᓕᕋᓗᐊᕐᑎᓗᒍ. 2011-ᒥ, ᐱᓇᓱᖃᑎᒌᑦ ᐊᖏᕈᑎᖃᕐᓯᒪᔪᑦ ᐅᓚᐱᖕᖏᓂᕐᓴᑯᑦ, ᑭᓯᐊᓂ ᐊᒥᓱᕕᓐᓂᓴᒃᑯᑦ ᖃᖓᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖃᖃᑦᑕᓂᒻᒧᑦ. ᑕᒪᓐᓇ ᓂᕈᐊᕐᑕᐅᓚᕿᓯᒪᔪᖅ ᓄᓇᖃᕐᖄᓯᒪᔪᑦ ᐃᓱᒫᓘᑎᖏᑦ ᑐᖕᖓᕕᒋᑦᓱᒋᑦ ᐊᐅᓚᑕᐅᓂᖏᓐᓅᓕᖓᔪᓄᑦ ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᐊᕐᓯᕈᓇᑐᒃᑰᒍᑎᐅᓰᑦ ᑮᓇᐅᔭᕐᑑᑏᑦ ᑕᑯᑦᓴᓚᕆᒃᑯᑦ ᖃᐅᓯᕐᓂᒨᓕᖓᔪᑦ ᐃᓂᖏᓐᓂᒃ. ᑕᒪᓐᓇ ᐱᓪᓗᒍ, 2011-ᒥ ᐱᒋᐊᕐᓱᒍ, ᖃᖓᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓃᑦ ᐊᑑᑎᔭᐅᖃᑦᑕᓯᒪᓕᕐᑐᑦ ᐊᕐᕌᒍᐃᑦ ᑕᓪᓕᒪᑕᒫᑦ. ᖃᐅᔨᓴᕐᓃᑦ ᐃᓚᖃᕐᓱᑎᒃ ᐅᖓᓯᓐᓂᖏᓐᓂᒃ ᐆᑦᑐᕋᐅᑎᖃᕐᓂᓂᒃ ᒪᕐᕈᕕᓪᓗᐊᑐᒥᓪᓗ ᖃᐅᔨᓴᕐᑎᓂᒃ ᐃᓂᖏᓐᓂᒃ ᖃᖓᑦᑕᔫᒃᑯᑦ ᓯᓈᖏᓐᓂ, ᐊᒻᒪᓗ 60-ᑭᓚᒦᑕᓄᑦ ᑎᑭᑦᓯᓂᒃ ᓄᓇᒥ ᕿᑭᕐᑕᓂᓪᓗ ᓯᓈᖏᓐᓂᐅᖏᑦᑐᖅ ᑕᓯᐅᔭᕐᔪᐊᒥ ᔦᒥᓯ ᐯᒥᓪᓗ. ᐋᕐᕿᓯᒪᐅᑎᖓ ᖃᓄᐃᓘᕈᓯᖏᓪᓗ ᓄᐃᑕᐅᓯᒪᔪᑦ ᒪᓕᑦᓯᒋᑦ ᖃᐅᔨᒪᐅᓰᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓯᕿᒥᖓᓂᒥᐅᓂ ᓇᓅᓐᓅᓕᖓᒧᓂ ᑐᕙᖃᕐᓇᖏᑐᒥ ᐱᔭᕆᐊᑐᔪᒃᑯᓗ ᖃᐅᔨᓴᕈᔩᑦ ᐊᑐᕐᑕᐅᓱᑎᒃ ᐆᑦᑐᕋᐅᑎᖃᑦᑖᕐᓂᒧᑦ ᐊᑦᔨᒌᖕᖏᑐᓂ ᓇᓄᖃᕐᓂᖏᓂ. ᓄᓇᒥ ᐱᓇᓱᑦᑐᑦ ᖃᐅᔨᓴᕐᓂᒥᒃ ᓯᕗᓪᓕᐹᖑᓗᓂ ᐱᔭᕇᕐᑕᐅᓯᒪᔪᖅ ᐋᓐᑎᐊᕆᐆᓯ 2011-ᖑᑎᓪᓗᒍ ᑯᐯᒻᒥᓗ 2012-ᒥ. ᑖᓐᓇ ᖃᐅᔨᓴᕐᓂᒥ ᖃᓄᐃᓘᕈᓯᖅ ᐊᑑᑎᒋᐊᓪᓚᑎᓪᓗᒍ 2016-ᒥ ᐋᓐᓂᐊᕆᐆᒥ ᑯᐯᒻᒥᓗ. ᐊᖑᒪᑦᓯᓂᕐᒧᑦ ᐊᐅᓚᑦᓯᓂᕐᒥ ᐱᔪᓐᓇᑎᑦᓰᑦ ᑐᕌᒐᓯᐊᕐᑕᖏᓐᓄᑦ, ᐅᐱᕐᖔᑯᑦ 2021-ᒥ ᖃᖓᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖃᓚᐅᔪᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓯᕿᓂᖓᓂ ᑕᕝᕙᓇᑦᓴᓇᖅ ᐊᕐᖁᖅᓯᑎᒃ ᓯᕗᓪᓕᒥ ᐊᑐᕐᑕᐅᓯᕕᓂᕐᓂ. ᑕᒪᓐᓇ ᑐᓴᕐᑎᓯᓂᖅ ᓴᕿᑎᑦᓯᕗᖅ ᐱᖓᓱᖏᓐᓂᒃ ᖃᐅᔨᓴᕐᓃᑦ ᓇᓪᓕᑳᕈᑎᑦᓴᖃᕐᓂᒧᑦ ᐃᓘᓐᓇᖏᓐᓂᒃ ᖃᐅᔨᓴᕐᓂᕕᓂᕐᓂᒃ (2011-12, 2016 ᐊᒻᒪᓗ 2021).

2021-ᒥ ᖃᖓᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᖃᖓᑕᓂᐅᓚᐅᔪᖅ ᐊᐅᒡᒍᓯ 22-ᒥᑦ ᓯᑎᐱᕆ 1-ᒧᑦ, ᑖᒃᑯᓂᖓᑦᓴᓇᖅ ᐊᑐᕐᓯᑎᒃ ᐋᕐᕿᓯᒪᐅᑎᖏᓐᓂᓗ ᐊᑑᑎᒍᑕᐅᓯᒪᔪᓂᒃ. 2021-ᒥ ᖃᖓᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᕐᓴᓂᖅ ᓄᐃᑦᓯᓚᐅᔪᖅ ᒪᕐᕈᐃᓂᒃ ᐃᒻᒪᖃᐅᑦᓵᕈᑏᓐᓂᒃ: 1) 1003 (95% CI 773-1302), ᓇᓪᓕᑳᕈᑕᐅᒍᓐᓇᓯᐊᕐᓯᓂ 2016-ᒥ ᖃᐅᔨᓴᐅᑎᕕᓂᒃᓄᑦ ᐅᓄᕐᓂᕆᒡᒐᔪᕐᑕᖏᓐᓂᒃ ᐊᒻᒪᓗ 2) 1119 (95% CI 860-1454), ᐃᒣᓪᓗᐊᒐᔪᕐᑐᓚᕆᐅᓂᕐᓴᓂᒃ ᓄᐃᑦᓯᓯᒪᔪᑦ ᐊᑐᕐᓯᑎᒃ ᓄᑖᖑᓂᕐᓴᓂᒃ ᑭᓯᑦᓯᒍᑎᓄᑦ ᖃᓄᐃᓘᕈᓯᕐᓂᒃ. ᑕᒪᕐᒥᒃ

ᐃᒣᓪᓗᐊᒐᔪᕐᑑᕈᑏᑦ ᓄᐃᑦᓯᓯᒪᔪᑦ ᐊᒥᓲᓂᕐᓴᓂᒃ ᓇᓄᕐᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓯᕿᓂᖓᓂ 2021-ᒥ 2016-ᖑᓯᒪᒻᒪᓂᑦ.

ᒪᖀᓄᓗᐊᖕᖑᐊᖅ ᐃᑲᔪᕐᓯᔭᐅᓯᒪᑐᐃᓐᓇᕆᐊᓖᑦ ᐅᓄᕐᓂᒋᐅᓕᕐᓂᖏᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓯᕿᓂᖓᓃᑦᑐᑦ ᓇᓄᐃᑦ ᐃᑉᐱᒋᔭᐅᓯᒪᔪᑦ 2021-ᒥ 2016-ᒥᓂᑌᑦ ᓴᓂᖏᓐᓂ: ᐊᕐᕌᒍᑕᓖᑦ ᖃᓄᐃᑦᑐᑐᐃᓐᓇᐅᓂᖏᑦ ᓄᓇᒥ ᐊᐅᓚᓂᖏᑦ ᓯᕿᓂᕐᒥᐅᑦ ᑕᓯᐅᔭᕐᔪᐊᒥ ᓯᓚᕐᖃᑎᒥᓐᓄᑦ ᓯᕿᓂᐅᑉ ᓂᐱᕕᐊᕐᓂᐅᓂᕐᓴᖅ ᑕᓯᐅᔭᕐᔪᐊᒥ, ᐱᐅᓂᕐᓴᐅᓕᕐᑎᓗᒋᑦ ᑕᑯᑦᓴᑦ ᐅᓄᕐᓂᖏᑦ. ᐃᒻᒪᖃ ᑕᒪᕐᒥᒃ ᐃᑲᔪᕐᓯᓯᒪᔪᑦ ᐃᑉᐱᒋᔭᐅᓂᖏᓐᓄᑦ ᐊᑦᔨᒌᖕᖏᓂᖏᑦᑕ ᐅᓄᕐᓂᖏᓐᓂ, ᐊᑦᔨᒌᖕᖏᓂᖏᓐᓄᑦ ᐳᕐᑐᓂᖏᑦᑕ, ᓲᕐᖁᐃᑐᒥᒃ ᑌᒣᓐᓂᕋᐅᑎᖃᖕᖏᑲᓗᐊᕐᓱᑎᒃ ᑲᒪᐅᑎᓂᒃ. ᓯᕿᓂᕐᒧᑦ ᑕᓯᐅᔭᕐᔪᐊᒥ ᓅᑦᑐᑕᓕᕕᓂᐅᔮᕐᑐᑦ ᓯᕿᓂᐅᑉ ᓂᐱᕕᐊᓂᑦ ᓇᓄᐃᑦ ᐃᓄᕐᓂᖏᓐᓂᒃ 2021-ᒥ ᖃᐅᔨᓴᕐᑕᐅᓂᕐᒦᓯᑎᒃ. ᐊᑦᔨᒌᖕᖏᓂᓚᓈᐱᓖᑦ ᐊᐅᓚᓂᖏᓐᓂ ᓇᓄᐃᑦ ᓯᕿᒥᖓᓂ ᓯᕿᓂᐅᓪᓗ ᓂᐱᕕᐊᕐᓂ ᑕᓯᐅᔭᕐᔪᐊᒥ ᓯᓈᓂ ᐅᓄᕐᓂᓴᕈᕐᑎᓯᓚᕿᑦᓯᑎᒃ ᓯᕿᓂᖓᓂᒥᐅᑕ ᕿᑐᕐᖏᐅᓯᒪᓂᒃᑯᑦ. ᐅᓄᕐᓂᓴᐅᓕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐃᒻᒪᖃ ᖃᐅᔨᓴᕐᑕᐅᓯᒪᔪᓂ 2021-ᒥ 2016 ᓴᓂᖏᓐᓂ ᑐᖁᕋᕐᑕᐅᖏᓐᓂᓴᐅᓐᓂᒪᑕ, ᕿᑐᕐᖏᓂᕐᓴᐅᓯᑎᒃ, ᐅᖅᑰᓗᓐᓃᑦ ᑕᒣᓐᓄᑦ. ᐊᕐᕌᒍᐃᑦ ᐱᖓᓱᑦ 2021-ᒥ ᖃᐅᔨᓴᕐᓂᕕᓂᕐᒥ ᐊᑯᓂᐅᓂᕐᐹᖅ ᑐᕙᖃᑦᓯᒪᔪᑦ ᑕᓯᐅᔭᕐᔪᐊᒥ ᐊᑑᑎᖃᑦᑕᓯᒪᔪᑦ ᑕᑯᓐᓇᓱᒋᑦ ᐊᕐᕌᒍᓂ ᖁᓕᓂ ᐊᓂᒍᕐᑐᓂ, ᓇᓐᓄᑕᐅᓂᖏᓪᓗ ᐃᑭᓐᓂᓴᐅᑦᓯᑎᒃ 2016-2021 ᐊᑯᓐᓂᖏᓐᓂ 2010 ᐊᒻᒪᓗ 2015 ᓴᓂᖏᓐᓂ; ᑕᒪᕐᒥᒃ ᖃᓄᐃᑦᑐᒦᓐᓃᒃ ᐅᓄᕐᓂᓴᕈᕐᓯᒪᓂᖏᓐᓄᑦ ᐱᑦᔪᑕᐅᕈᓐᓇᑐᑦ.

ᐃᑉᐱᒋᔭᐅᓂᖏᑦ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦᑕ ᓯᓚ ᐊᓯᑦᔨᐸᓪᓕᐊᓂᖃᕐᑎᓗᒍ ᓄᓇᓕᒻᒥᐅᓂᒃ ᐃᑲᔪᕐᓯᓗᐊᕐᖃᔭᖕᖏᑎᑦᓯᕗᖅ ᑐᓴᐅᒪᔭᐅᒋᐊᓪᓛᕆᐊᓕᓐᓄᑦ ᐆᒪᔪᕐᓂᐊᓂᕐᒧᑦ ᑐᑭᑖᕈᑎᖃᕈᓐᓇᓂᕐᒧᑦ. ᐃᑉᐱᒋᔭᐅᒍᑎᙯᓐᓇᓂᖏᑦ ᕿᑐᕐᖏᐅᓂᖏᑦᑕ, ᐊᓐᓇᐅᒪᒍᓐᓇᓂᖏᑦᑕ, ᐊᕐᕌᒍᒥᓪᓗ ᐊᐅᓚᓂᓇᓲᖏᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓯᕿᓂᐅᑉ ᓂᐱᕕᐊᓂ ᓯᕿᓂᖓᓂᓪᓗ ᐱᓪᓚᕆᐅᓚᖓᕗᑦ ᑐᓴᐅᒣᓐᓇᑎᑦᓯᒍᑎᓄᑦ ᐊᐅᓚᑦᓯᓂᓕᕆᔨᓂᒃ ᐊᑯᓐᓂᖏᓐᓂ ᖃᖕᒐᑦᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᓂ.

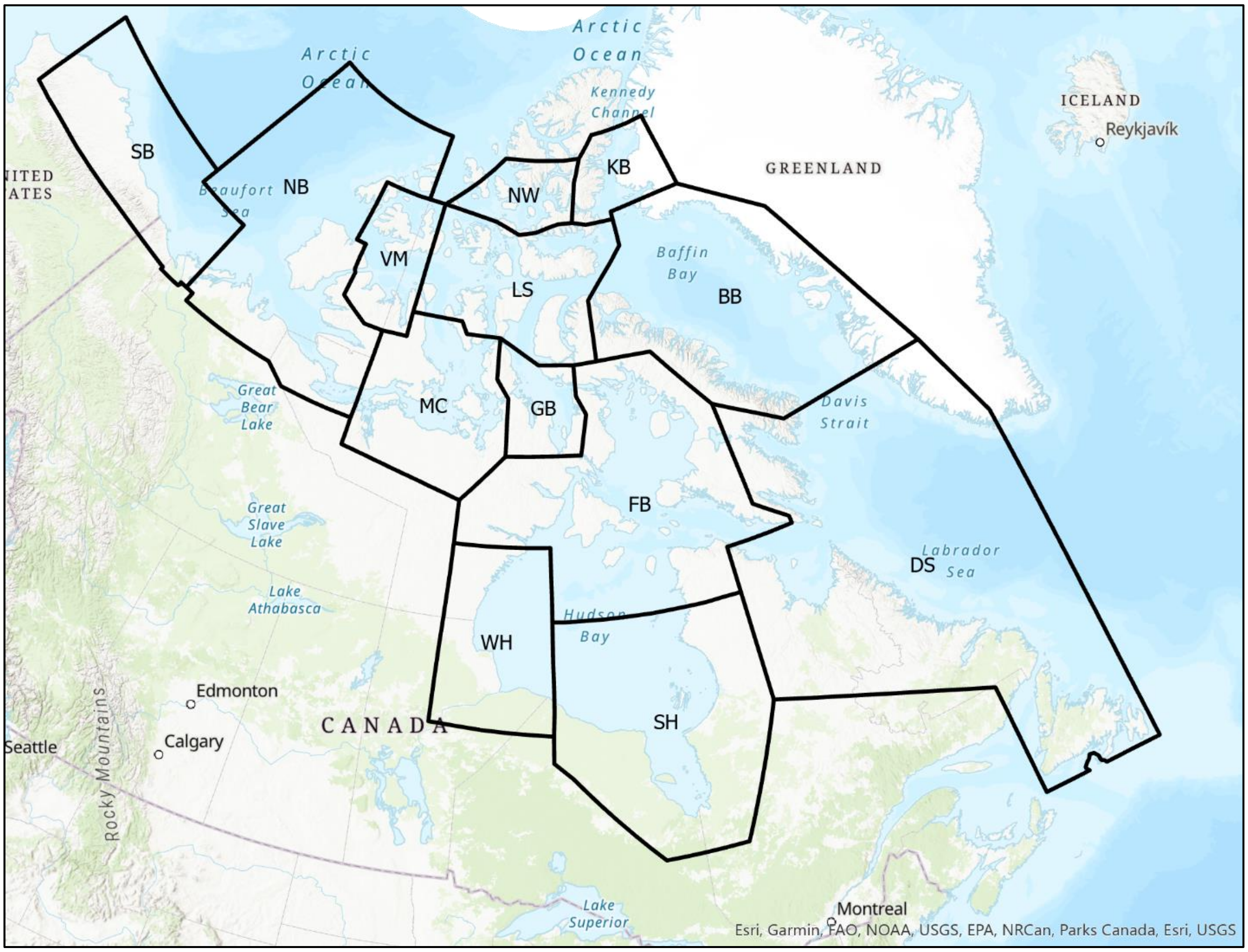
INTRODUCTION

Climate warming is causing rapid and widespread impacts to Arctic ecosystems (Post et al. 2009) where temperatures are increasing at two to four times the global average (IPCC 2013, Rantanen et al. 2022). These impacts have had profound effects on a variety of Arctic wildlife species, causing population declines, reduced reproductive output, and shifts in the food web (Regehr et al. 2007, Post and Forchhammer 2008, Laidre et al. 2015, Descamps et al. 2017, Mallory and Boyce 2018). The impacts of climate change on Arctic ecosystems have had significant consequences for Indigenous peoples that rely on Arctic species for subsistence (Durkalec et al. 2015, Laidre et al. 2015, Ostapchuk et al. 2015, Kanatami 2019). As climate change continues to alter Arctic ecosystems (IPCC 2022), it is critical to monitor impacted species to provide information to local communities for use in decision-making and to assess general impacts to people and biodiversity from a warming climate.

Polar bears (*Ursus maritimus*) exemplify the challenges facing Arctic species under a changing climate. Polar bears are dependent on sea ice for nearly every stage of their life: they hunt their primary prey from the sea-ice platform, mate and, in some locations, even den on the sea ice (Amstrup and Gardner 1994). Thus, declines in sea ice have direct implications for nutrition, reproduction and the long-term population viability for polar bears. Although sea-ice extent and duration have declined in the last few decades over the circumpolar distribution of polar bears (Stern and Laidre 2016), the impacts to polar bear subpopulations have varied, with some experiencing declines in body condition, survival and abundance (Regehr et al. 2007, Lunn et al. 2016, Obbard et al. 2016, Obbard et al. 2018) and others experiencing limited effects or even near-term benefits as areas transition from multi-year ice to thinner, annual ice or areas in which access to shallow, highly productive ecoregions remains (Regehr et al. 2018, Laidre et al. 2020, Dyck et al. 2021, Dyck et al. 2022).

Polar bears are an important cultural, nutritional and financial species to Indigenous peoples that have coexisted with them for centuries (Wenzel 2004, Henri et al. 2010, Laforest et al. 2018).The harvest of polar bears is monitored through management frameworks in various jurisdictions across Canada (Taylor et al. 2008, Lunn et al. 2018), all aiming for sustainable harvest management and continued population viability. However, the logistical and analytical challenges involved with enumerating polar bear populations, as well as the often long intervals between surveys, adds uncertainty to the achievement of this goal. Compounding uncertainty of the responses of bears to climate warming increases the complexity of identifying the sustainability of harvest levels (Regehr et al. 2017, Regehr et al. 2021). Thus, monitoring polar bear populations in the face of ongoing climate warming is critical for providing local communities that rely on polar bears with additional information for harvest management decision-making.

Polar bears are divided into 19 relatively discrete subpopulations (Durner et al. 2018) delineated using a variety of methods, including capture and recapture data, genetics, and movement data from collared individuals (Paetkau et al. 1999, Taylor et al. 2001, Amstrup et al. 2004). The Southern Hudson Bay (SH) subpopulation represents the furthest south continuously occupied area of the globe for polar bears, and, as such, is a critical location for monitoring the impacts of climate warming. The marine portions of the SH subpopulation include the eastern and southern portions of Hudson Bay and all of James Bay (Fig. 1). The subpopulation also encompasses nearly the entirety of the coastline of Ontario, large areas of the western coastline of Québec, and areas of both provinces up to 120 km inland.



Esri, Garmin, FAO, NOAA, USGS, EPA, NRCan, Parks Canada, Esri, USGS

Figure 1. Boundaries of polar bear subpopulations that are partially or totally under management by Canadian jurisdictions. SB, Southern Beaufort Sea; NB, Northern Beaufort Sea; VM, Viscount Melville Sound; MC, M'Clintock Channel; LS, Lancaster Sound; NW, Norwegian Bay; KB, Kane Basin; BB, Baffin Bay; GB, Gulf of Boothia; FB, Foxe Basin; DS, Davis Strait; WH, Western Hudson Bay; and, SH, Southern Hudson Bay.

The first abundance estimate for SH was obtained between 1984 and 1986 by Kolenosky et al. (1992) using physical capture-mark-recapture conducted primarily along the Ontario coast of Hudson Bay and including some inland areas. This effort extended somewhat into the current limit of the Western Hudson Bay (WH) subpopulation and produced an estimate of 763 bears (± 323) but was later adjusted upwards to 1000 bears for management purposes because no sampling was conducted on the James Bay coast of Ontario, the Québec coast, or any of the offshore islands of James and Hudson bays (Lunn et al. 1998). During 1997 and 1998, a capture-mark-recapture effort was undertaken on Akimiski, North and South Twin Islands in James Bay. Although a formal estimate was never published for these efforts, Obbard et al. (2007) citing Obbard and Howe (unpublished data) report abundance estimates ranging from 70 to 110 bears, which were derived from several models (minimum lower confidence limit across models = 56 and maximum upper confidence limit across models = 195). Between 2003 and 2005, Obbard et al. (2007) conducted another physical capture-mark-recapture effort, covering the same area as assessed in the 1980s, but more thoroughly covering areas up to 40 km inland from the coast. Further, they reanalyzed the data from 1984-1986 excluding captures occurring outside of the current SH subpopulation boundary. This work estimated that there was an average of 641 bears (95% CI = 401-881) between 1984 and 1986 and 681 bears (95% CI = 401-961) between 2003 and 2005 in the study area, indicating the population in the surveyed area was likely very similar between the two survey periods. However, concurrent with these abundance estimates, declines in the point estimates of survival between the 1980s and 2000s were documented (Obbard et al. 2007) as well as significant declines in body condition of bears (Obbard et al. 2016). Further, the ice-free season in SH increased by approximately three weeks between the 1990s and 2010s (Hochheim and Barber 2014). Thus, while it appears that the population abundance along the Ontario coast of Hudson Bay and the areas inland was largely similar between the 1980s and mid-2000s, there was evidence that the population might be facing nutritional issues and attendant declines in survival and body condition related to declining sea ice. Concurrently, the adjacent WH subpopulation had seen similar declines in survival and body condition as well as abundance during the same period (Regehr et al. 2007, Lunn et al. 2016). Lastly, there remained areas of the subpopulation, including the Québec coast, large portions of the James Bay coast, and several James Bay and Hudson Bay islands, that had still not been surveyed rigorously enough to contribute to abundance estimates at that point (Leafloor 1990, Crête et al. 1991).

Although physical capture programs offer some of the best data for understanding polar bear vital rates and population dynamics, while also enabling the collection of data on body condition, they are logistically challenging, expensive to undertake, and take several years to produce robust estimates. Further, Indigenous peoples that coexist with polar bears have raised concerns about the handling and chemical immobilization of polar bears for scientific and management purposes (Peacock et al. 2009, Service Canadien de la Faune 2010, Henri et al. 2010, Wong et al. 2017, https://www.itk.ca/wp-content/uploads/2019/08/A09-06-11-Approval-of-Polar-Bear-Research-Methods.pdf accessed November 16, 2022). Starting in 2011, management authorities for SH and WH moved to an aerial survey-based approach for enumerating these subpopulations (Stapleton et al. 2014, Obbard et al. 2015, Dyck et al. 2017). Less information is gained through aerial surveys relative to mark-recapture efforts, so, after conducting power analyses, jurisdictions agreed that surveys would occur on a more regular basis and be repeated every five years. Thus, in 2011, Obbard et al. (2016) implemented a combined distance sampling and double-observer mark-recapture aerial survey of the Ontario coast and areas up to 60 km inland along with Akimiski Island. At the time, there was insufficient funding to also Survey the Québec coast and offshore islands of James and Eastern Hudson Bay (M. Obbard personal communication), but these areas were subsequently surveyed in 2012. This was the most comprehensive survey of the SH subpopulation to date and produced an estimate of 943 bears (95% CI = 658-1350). This survey was repeated in 2016, with all areas surveyed in a single season (Obbard et al, 2018). This effort produced an estimate of 780 bears (95% CI = 590-1029), suggesting the population may have declined between 2011 and 2016. Further, the age composition of observed bears in the 2016 survey was suggestive of a poor survival of cubs to yearling stage considering few yearling bears were seen. An additional double-observer mark-recapture survey of only the coastline of Ontario, where the greatest density of bears occurs, was conducted in 2018 to examine indices of recruitment and obtain an estimate of the coastal population. This survey was an exact replicate of a portion of the 2011 and 2016 double-observer mark-recapture surveys, which allowed for a direct comparison of this portion of the population across years. The results showed that the proportion of yearlings was slightly higher in this area in 2018 than in 2016, but the number of bears inhabiting the coast was slightly lower at 249 bears (95% CI = 230-270) compared to 2016 ($\bar{x}$ = 269, 95% CI = 214-297) and substantially lower than 2011 ($\bar{x}$ = 422, 95% CI = 381-467; Northrup and Howe 2019).

Similar to other subpopulations in Canada, the harvest of SH polar bears has long been targeted for a 4.5% removal rate at a sex ratio of 2 males per female. This rate has been considered sustainable for polar bears (Taylor et al. 1987), though there is evidence that it may have been conservative for bears in SH over the last 20 years (Regehr et al. 2021). Polar bears in the SH subpopulation are harvested by Inuit in Nunavut and Nunavik and by Cree in Québec and Ontario, though recorded Cree harvests in Ontario were much greater in the 1970s through 1990s than at the time of this report (OMNRF unpublished data). Management authority for the SH subpopulation is complex as it is the shared responsibility of the Governments of Ontario, Québec, Nunavut, and Canada, along with the Nunavut Wildlife Management Board, Nunavik Marine Region Wildlife Management Board, the Eeyou Marine Region Wildlife Board, Hunting, Fishing and Trapping Coordinating Committee, Land Claims Organizations representing Indigenous rights, specifically Nunavut Tunngavik Incorporated, Makivik Corporation and the Cree Nation Government in Québec, and several Cree First Nations in Ontario. The harvest of SH bears in Nunavut has been managed under a strict quota system since the 1970s, whereas harvest monitoring in Québec and Ontario remains incomplete as of this report. Total annual reported harvest within the subpopulation varies annually but averaged 48 bears between 2010-11 and 2020-21 (range 31-104; https://www.polarbearscanada.ca/en/polar-bears-canada/canadas-polar-bear-subpopulations; accessed July 22, 2022).

There are sixteen coastal communities in the SH subpopulation (Fig. 2). Between 1980 and 2019, the Inuit community of Sanikiluaq, Nunavut had a total allowable harvest (TAH) of 25 bears at a male to female ratio of 2:1. The Sanikiluaq harvest was reduced to 20 bears per year for two years following the 2011-12 aerial survey. The management framework allows for annual variation in the actual harvest depending on over- or under-harvest compared to the TAH (Government of Nunavut 2019). A revision of the Nunavut polar bear harvest management system in 2019 allows the sex ratio of the harvest to reach up to one female bear for every male bear (up to 1:1). With this management change, the TAH for Sanikiluaq remained at 25 bears, indicating the potential for a greater number of female bears to be harvested after this time. Harvest reporting in Nunavut is believed to approach 100% and the average annual reported harvest for the 2010-11 to 2020-21 period was 26.2 bears (range 20 to 47 bears).

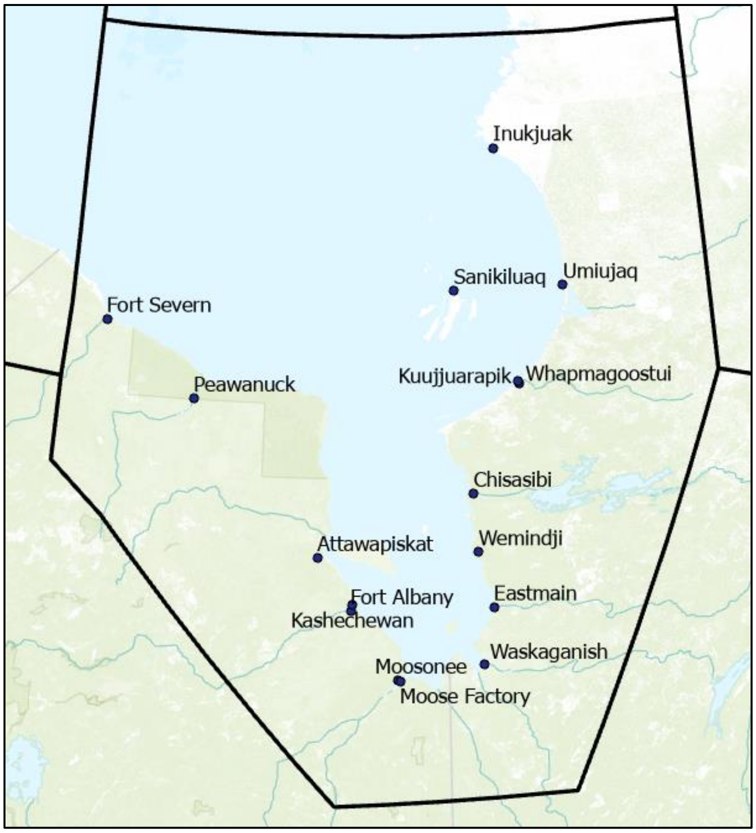


Figure 2. Coastal communities falling within the SH subpopulation boundary in Ontario, Québec and Nunavut.

In Québec, three Nunavik Inuit communities (Inukjuak, Umiujaq, and Kuujjuaraapik) and five coastal Cree communities (Whapmagoostui, Chisasibi, Wemindji, Eastmain and Waskaganish) potentially harvest from this subpopulation. There are currently no legal requirements for beneficiaries of the James Bay and Northern Québec Agreement (Québec Government 1976) to report human-caused polar bear mortalities but reporting and tagging of polar bear hides is necessary for hides to enter the domestic or international trade market. The proportion of the harvest reported to the Québec Government is currently unknown. Voluntary agreements were signed in 2011[1] and 2014[2] establishing harvest limits within the SH subpopulation for Nunavik Inuit and Cree of Eeyou Istchee and Ontario, and a total allowable take (TAT) was also established by the federal and Nunavut governments in 2016 for bears harvested within the Nunavik Marine

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[1] A temporary voluntary limit of 26 bears for Nunavik Inuit, 25 for Inuit from Sanikiluaq, 4 for Cree of Eeyou Istchee, and 5 for Ontario Cree was established (including subsistence hunting and defense kills) for the 2011/12 harvest season.

[2] A temporary voluntary limit of 22 bears for Nunavik Inuit, 20 for Inuit from Sanikiluaq, and 3 bears for Ontario and Québec Cree with alternating division per harvest season for Cree was established for the 2014/15 and 2015/16 harvest seasons.

Region[3]. However, enforcement of those harvest limits remains problematic, and no harvest limits have been established in most of the Eeyou Marine Region nor in onshore Québec. Average annual reported harvest in Québec for the 2010-11 to 2020-21 period was 19.7 bears (range 5 to 74 bears).

In Ontario, there are three coastal Cree communities that have traditionally harvested polar bears (Fort Severn, Winisk (Peawanuck) and Attawapiskat). There are three additional Cree communities (Moose Factory, Fort Albany, and Kashechewan), and one non-Indigenous community (Moosonee) that are outside the generally occupied range of bears but occasionally have defense of life and property kills. In 1976, an informal agreement between the Ontario government and the coastal Cree First Nation Communities established that a maximum of 30 bear hides could be sealed for trade annually. The 2011[4] and 2014[2] voluntary agreements also set maximum harvest limits on Ontario Cree but the proportion of the harvest that is reported to the Government of Ontario is currently unknown. Since polar bears were listed as threatened in Ontario in 2009, the sale of bear parts has been prohibited in the province.

A harvest risk assessment conducted by Regehr et al. (2021) indicated that under ongoing climate warming, harvest of polar bears in SH would likely need to decline in coming years to ensure harvest sustainability. Further, evidence outlined above suggests the SH subpopulation may be experiencing demographic challenges related to ongoing declines of sea ice. As such, there is a clear, continued need to assess the abundance of this subpopulation to monitor trend and support harvest management (Regehr et al. 2021). In keeping with management authority goals, a comprehensive aerial survey of SH was conducted in summer 2021 that maintained a nearly identical design as the previous surveys. Here we present the results of this third survey to provide a direct comparison across the three survey periods (2011/12, 2016 and 2021).

METHODS

Study area

The survey area was established according to the known distribution of SH bears during the ice-free season (Prevett and Kolenosky 1982, Obbard and Middel 2012). This area is large, topographically and vegetatively diverse, and has high variability in polar bear

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[3] A harvest limit of 23 bears within the Nunavik Marine Region was established for Nunavik Inuit, with at least one tag allocated to the Cree of Eeyou Istchee for harvest within the Inuit-Cree overlap area.

[4] A temporary voluntary limit of 5 bears was established for the six coastal Cree Nations of Ontario (including subsistence hunting and defense kills) for the 2011/12 harvest season. Not all Ontario communities were included in discussion about this voluntary limit.

density. It spans large portions of the northern Ontario and northern Québec coasts and inland areas, with the islands of James Bay and Hudson Bay being part of the Territory of Nunavut (Fig. 1 and 2). The Ontario portions of the subpopulation are part of the Hudson Bay lowlands ecosystem, consisting of large wetland complexes, extensive treed areas and tundra along the coast of Hudson Bay (Fig. 3). This area has little topographic relief and the coastal portions include extensive tidal flats (Fig. 3). The Québec portion of the study area consists of a series of long and steep rocky nearshore islands forming the Nastapoka Island complex as well as a relatively flat and hilly shrub tundra shoreline. The subpopulation also includes a large number of islands in James and Hudson bays, including the large Akimiski Island, the Twin Islands and the Ottawa islands complex that are known to be used extensively by polar bears during the ice free season. Southeastern Hudson Bay also holds the Belcher islands archipelago spreading over almost 3000 km^2. There are numerous Cree and Inuit communities along the Ontario and Québec coast and one Inuit community on the Belcher Islands.

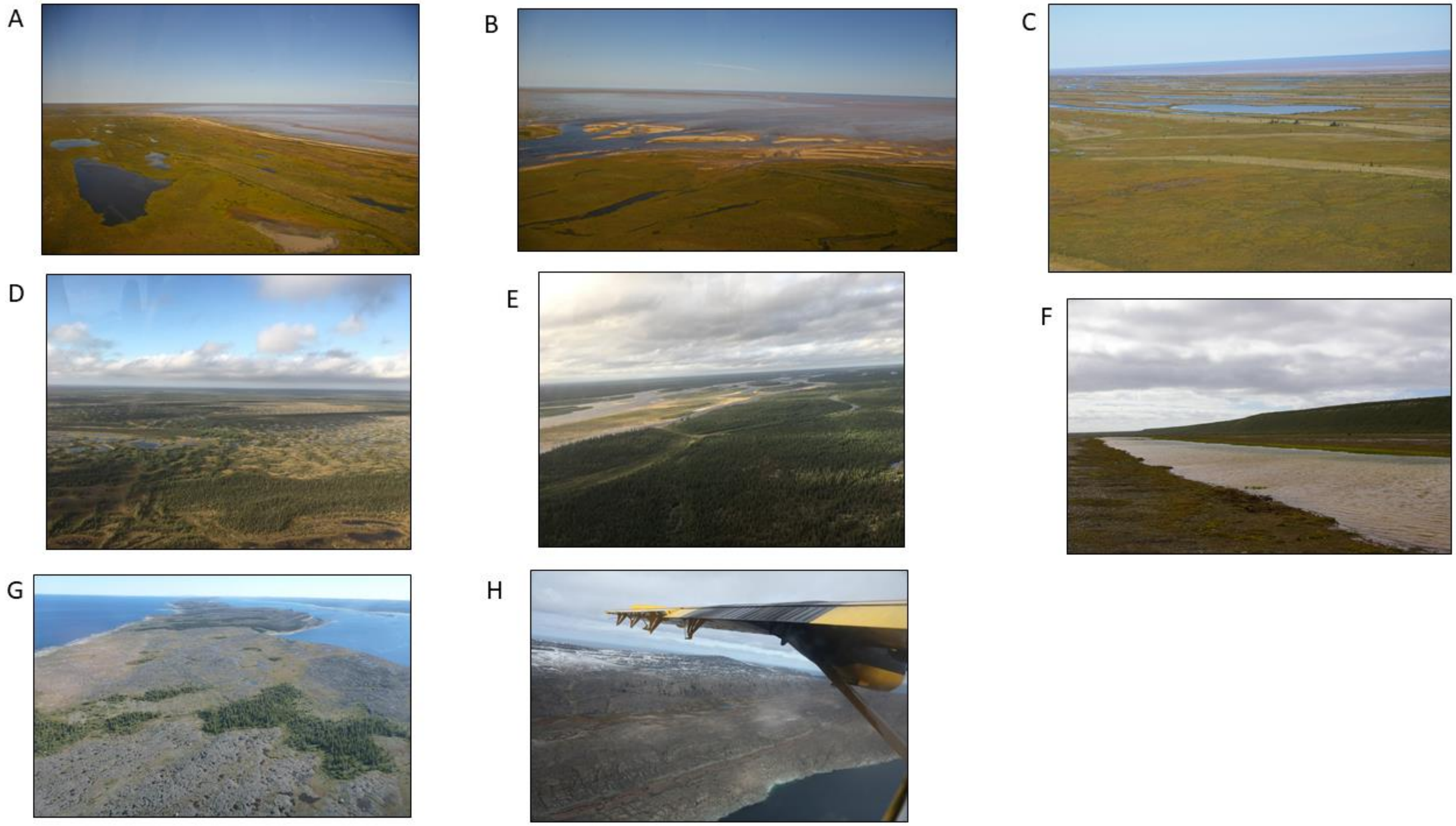


Figure 3. Representative photos of the vegetation and topography of the SH subpopulation. (A) The majority of the Hudson Bay coastline in Ontario consists of open tundra with interspersed wetlands and dry beach ridges. (B) There are extensive mudflats throughout the entirety of the Ontario coastal area. (C) Further inland from the Hudson Bay coast of Ontario is a mix of dry beach ridges, open tundra and wetlands. (D) Further inland from the Hudson Bay coast of Ontario and throughout most of the inland areas of James Bay there are interspersed treed areas, palsas and wetlands. (E) eventually, these areas give way to extensive treed areas and large riverine systems. (F) The islands of James Bay contain substantially more topography than the mainland Ontario portion of the study area. Shown here is North Twin Island. (G) The Québec coastline of James Bay is likewise more topographically diverse and consists of numerous small rocky islands. (H) Hudson Bay has numerous rocky islands where bears summer. Shown here is a portion of the Ottawa Islands.

Survey design

We followed the survey design implemented in 2011/12 and 2016 (Obbard et al. 2015, Obbard et al. 2018) to provide a comparable population estimate. The 2011 and 2012 surveys were designed based on scientific information on the distribution of bears in SH during the ice-free season and information obtained from consultation with Indigenous communities in the region. Following the 2012 survey, a second round of consultation was conducted in Québec to address points raised by Inuit communities and Makivik Corporation. This resulted in the addition of a series of inland transects perpendicular to the Québec coast along with a few additional islands in James Bay to the design of the 2016 survey to fully represent the scientific and Inuit knowledge of bear distribution in the area during the ice-free season. The surveys leverage the fact that Hudson Bay is entirely ice-free from approximately early August to late November each year during which time bears in SH are onshore. Further, females do not enter dens until October and November (Middel 2014), thus, between mid-August and the end of September, all bears are accessible (onshore) and available to be surveyed. We surveyed the subpopulation during this time and as close as possible to a similar survey being conducted in adjacent WH aimed to mirror the 2011 and 2016 WH surveys (Atkinson et al. 2022). As in past surveys (Obbard et al. 2015, Obbard et al. 2018), we subdivided the study area into regions based on expected bear density, aircraft type and survey design (Fig. 4). Past research has shown that the majority of bears in this subpopulation spend the ice-free season on the Ontario mainland, with a at least 10% of the population also inhabiting the islands of James Bay and eastern Hudson Bay (Obbard et al. 2015, Obbard et al. 2018). Although bears are regularly observed during winter along the Québec coast of Hudson Bay, bears are rare in that part of their range during the summer and are mostly sighted on Long Island and the Cape Jones area (Nunavik Marine Region Wildlife Board [NMRWB] 2018). This was also confirmed by the surveys in 2012 and 2016, which failed to observe any bears along the Québec coastline or inshore (Obbard et al. 2015, Obbard et al. 2018). Thus, we divided the study area into 1) the Ontario mainland, coastline, and Akimiski Island, located in James Bay, 2) the James Bay and Hudson Bay islands, excluding Akimiski Island, 3) nearshore islands off the Ontario coast and 4) the Québec coastline and nearshore islands (Fig. 4). Note that below, we aimed to refer to these areas exactly as they are listed above whenever mentioned to reduce confusion due to the complex nature of the study design.

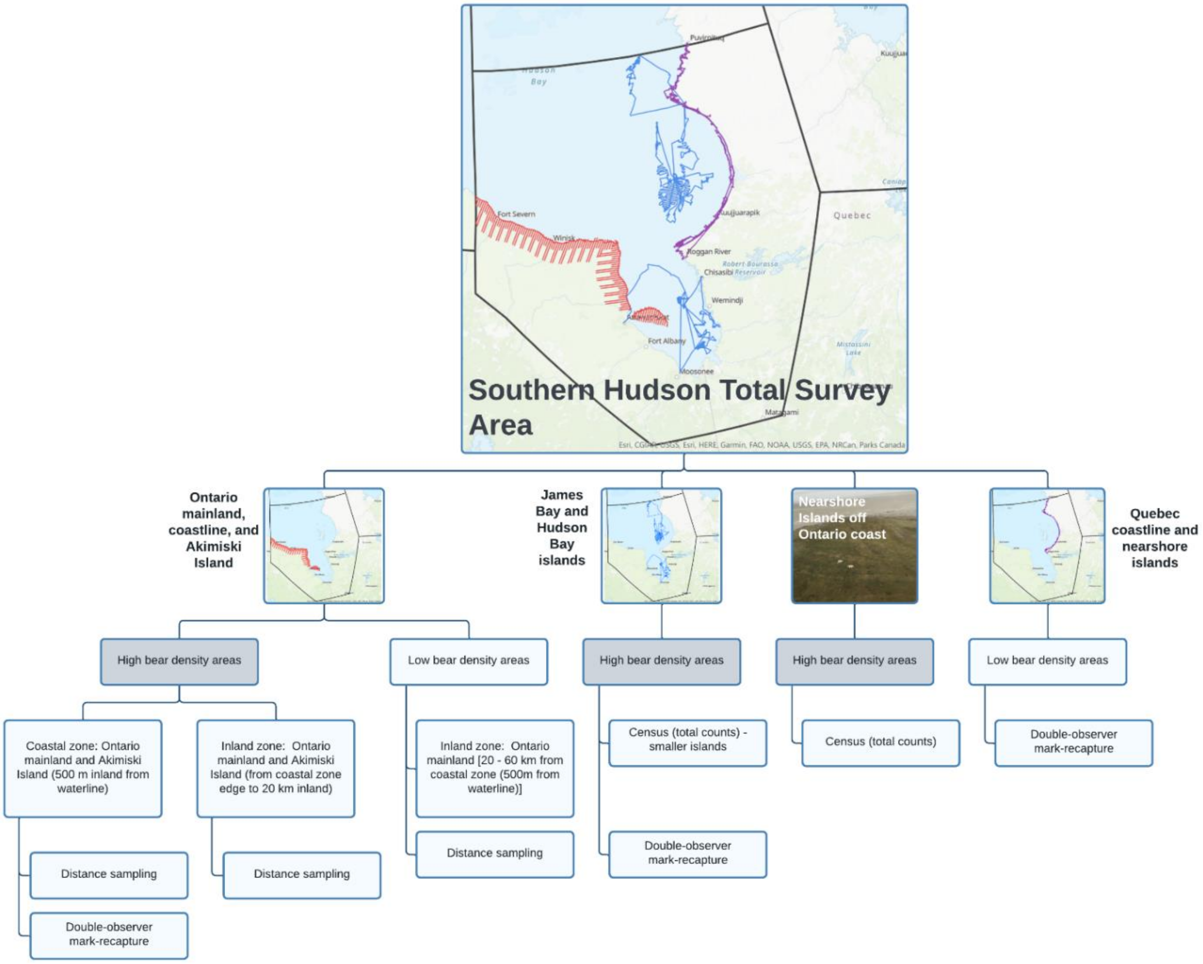


Figure 4. Schematic outlining the different survey areas, designs and analytical techniques used in SH polar bear survey in 2021.

Ontario mainland, coastline, and Akimiski Island

Most of the bears within the SH subpopulation summer on the Ontario mainland, with the majority of these bears concentrated along the coast (Kolenosky et al. 1992, Obbard and Middel 2012, Middel 2014, Obbard et al. 2015, Obbard et al. 2018). However, bears are also regularly documented far inland. Akimiski Island historically has held a high density of bears (Obbard et al. 2007), is only a short distance from mainland Ontario and is reachable via single-engine helicopter. Thus, it was surveyed in an identical manner to the Ontario mainland. We subdivided the Ontario mainland, coastline and Akimiski Island into 2 strata (Fig. 5). We designated areas from 20 km inland out to the waterline, including exposed mudflats, and the entirety of Akimiski Island as the high-density stratum. We designated all areas between 20 km and 60 km inland as the low-density stratum. Although bears have been documented further than 60 km inland (Kolenosky et al. 1992, Lemelin et al. 2010), such occurrences appear to be relatively rare, and the timing of the survey was such that pregnant females would not yet have entered their dens, which can occur far inland. Once the high-density stratum area was delineated, we further subdivided it into a coastal zone and inland zone (Figs. 4 and 6). The coastal zone consisted of all areas 500 m inland from the approximate high-tide line out to the waterline. Depending on when these areas were flown relative to high tide, this coastal zone could consist of large expanses of mud flats and numerous spits. The inland zone of the high-density stratum was all areas from 500 m inland from the approximate high-tide line to 20 km inland.

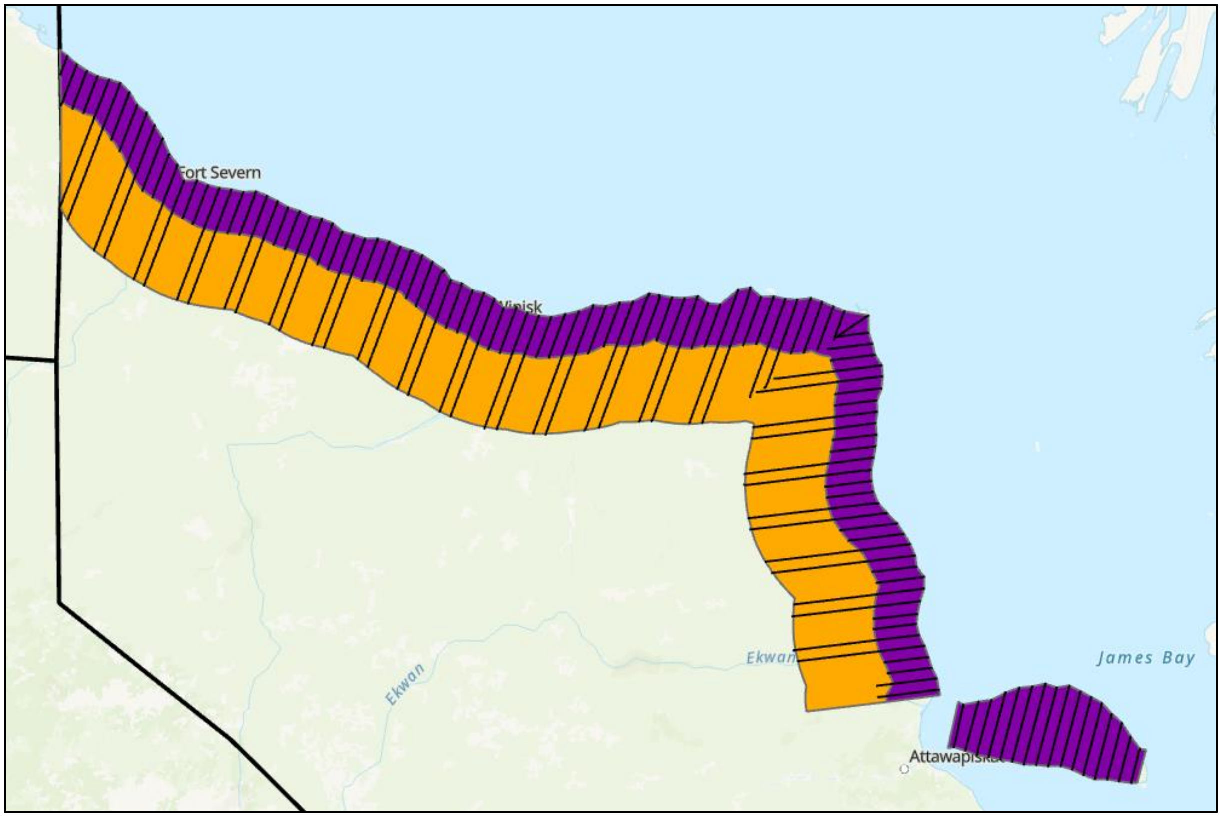


Figure 5. Flight lines (black lines) and stratum delineation for distance sampling survey of Ontario mainland, coastline and Akimiski Island. Purple shading represents the high-density stratum, consisting of all areas of mainland Ontario within 20 km of the waterline as well as the entirety of Akimiski Island. Orange shading represents the low-density stratum, consisting of all areas between 20 and 60 km from the waterline.

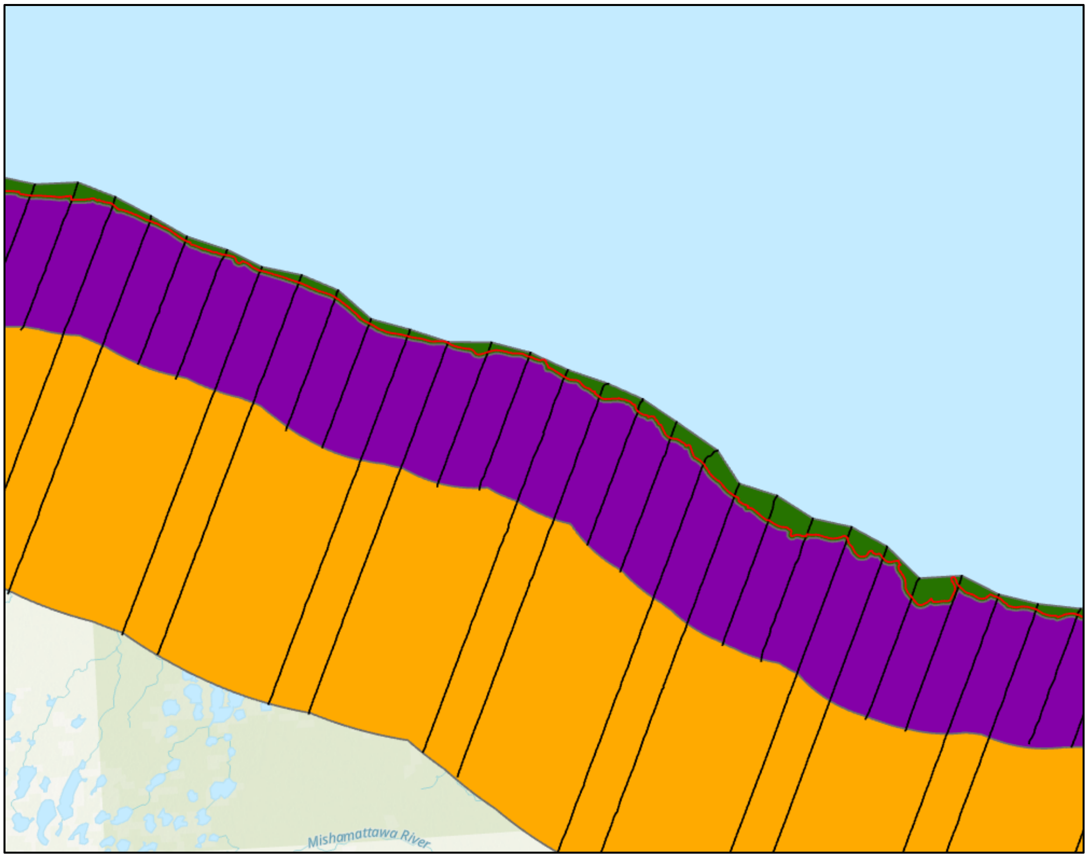


Figure 6. Close-up example of the delineation of the Ontario mainland, coastline and Akimiski Island area into different strata and survey approaches. Purple shading represents the inland zone of the high-density stratum, consisting of all areas of mainland Ontario between 20 km and 500 m from the approximate high-tide line, and the entirety of Akimiski Island further than 500 m from the approximate high-tide line. The green shading represents the coastal zone of the high-density stratum, consisting of all areas from 500 m inland from the high-tide line to the waterline. Orange shading represents the low-density stratum, consisting of all areas between 20 and 60 km from the approximate high-tide line. Red line represents the flight line for the double-observer mark-recapture portion of the survey.

Based on the above, the Ontario mainland, coastline and Akimiski Island area consisted of 3 sub-areas: 1) the coastal zone of the high-density stratum, 2) the inland zone of the high-density stratum, 3) the low-density stratum (Fig. 4 and 6). We employed two different survey techniques within these areas to address the strong variation in bear density among them. First, we employed a mark-recapture distance sampling survey covering the entirety of both the low and high-density stratum (i.e., both the inland and coastal zones in the high-density stratum). Following past surveys (Obbard et al. 2015, Obbard et al. 2018), transects were spaced 6 km apart across the entire high-density stratum including Akimiski Island (Fig. 5). Every other pair of transects was extended into the low-density stratum such that the low-density stratum was flown using pairs of transects spaced 6 km apart with the pairs separated by 18 km (Figs. 5 and 6). When present, these transects were extended out over exposed mudflats. If transects coincided with the small nearshore islands (see below) known to hold large numbers of bears, they were truncated at these islands to exclude the islands from our distance sampling estimate because these were surveyed separately as described below in section: *Nearshore islands off Ontario coast.*

For all three survey areas of the Ontario mainland, coastline and Akimiski Island, we employed distance sampling, flying transects in a Eurocopter EC-130 helicopter at an altitude of 120 m above ground level (AGL) and a speed of 160 km/h between August 22 and September 1, 2021. The crew consisted of a pilot, navigator (front right side of helicopter) and two rear observers positioned behind the pilot and navigator. All four, including the pilot, scanned for bears. Throughout the survey, the same pilot and observers participated, and all maintained the same position in the helicopter. We erected an opaque barrier between the front and rear of the helicopter to ensure rear observers were not alerted to the presence of a bear by the movements of the front observers. Further, observers allowed sufficient time from first detection of a bear for the other observers to have detected it. Once sufficient time had elapsed, it was determined whether the front observer, rear observer or both had detected the bear. We then flew to the approximate location of where the bear was first spotted and recorded a GPS location for calculating distance from the transect line. We recorded the position of who had observed the bear (pilot only, navigator only, back right only, back left only, both observers on the left or both observers on the right), the age class and sex of the bear (adult male, lone adult female, subadult, female with cubs of the year, female with yearlings), the group size, including all dependent offspring, the body condition on a 5 point scale (5 obese, 4 above average, 3 average, 2 below average and 1 emaciated), the activity of the bear when first spotted, the general habitat where the bear was first seen (e.g., mudflat or forest), a 3 point subjective scale for visibility, the general weather, vegetation height and density surrounding the bear, each on a 3 point scale, the degree to which glare from the sun was impacting visibility on a subjective 3 point scale and lastly, whether the bear was positioned relative to the helicopter such that it was unavailable to be observed by the rear observers (i.e., was in the rear observers' "blind-spot"). The availability of the bear to be observed by rear observers was reduced for bears near the transect line, but the exact distance varied depending on the orientation of the helicopter. In crosswind conditions, the helicopter often was "crabbing" and not oriented in the same direction as the transect line (Fig. 7) which influenced the position and width of the blind-spot. Finally, if another bear was observed while collecting covariate information off the transect line, it was not included in detections as it was assumed to have not been detected from the transect line.

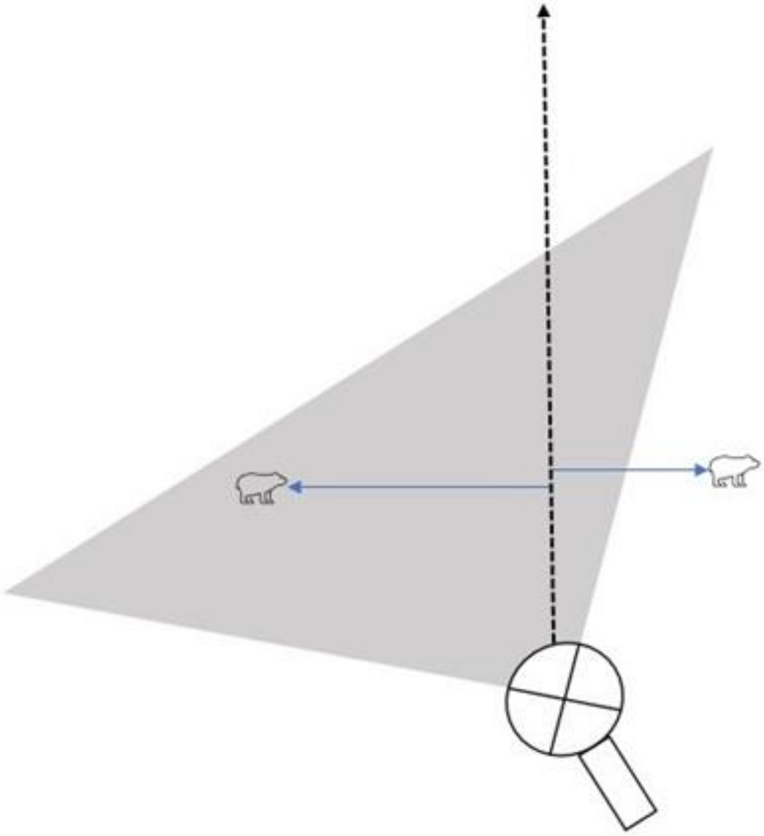


Figure 7. Schematic showing the influence of the orientation of the helicopter relative to the flight line on the ability of rear observers to observe bears on and close to the transect line. In this schematic, the dashed line represents the transect line and the gray polygon the blind-spot for rear observers. In this example, because the helicopter was oriented at an angle relative to the transect line, bears would be observable closer to the transect line for the right rear observer than the left rear observer.

In addition to the distance sampling survey, we also conducted a double-observer mark-recapture survey covering the coastal zone of the high-density stratum (i.e., the area within 500 m of the high-tide line extending out to the water line). We flew parallel to the coast at the approximate high-tide line and recorded detections of bears within 500 m inland and out to the waterline, including exposed mudflats. Observer setup within the helicopter, flight speeds, and recorded covariates were as described above. The use of both distance sampling and mark-recapture survey methodologies results in the coastal zone being sampled twice: once during the mark-recapture survey where we flew parallel to the coast and once during mark-recapture distance sampling where transects were flown perpendicular to the coast. Use of both surveys to obtain an averaged estimate (Obbard et al. 2015, Obbard et al. 2018) makes the assumption that bear position within the coastal zone is constant. Although movement of bears due to the helicopter generally appears only slight, the coastal zone is narrow and thus the estimate would be subject to fluctuation from bears moving into or out of the zone due to the helicopter. Thus, we attempted to fly the coastal zone mark-recapture survey on the same day, but prior to the overlapping distance sampling transects. Because the coastal zone is part of the high-density stratum, which extended an additional 19.5 km inland from the edge of the coastal zone, slight movements into or out of the coastal zone do not affect our distance sampling estimate. A large number of bears would need to move >20 km in a short period of time in response to the helicopter for bias to occur.

James Bay and Hudson Bay offshore islands

The James Bay and Hudson Bay Islands were considered high bear density areas and surveyed between September 2nd and September 10th, using double-observer mark-recapture from a de Havilland DHC-6 Twin Otter airplane. The coverage was identical to the area surveyed in the 2016 study. We flew at an average altitude of 150 m AGL and at a target speed of 150 km/h. The shape, size, and topography of the islands in James and Hudson Bays required variable flight patterns to ensure comprehensive coverage. We surveyed the Belcher Islands complex in Hudson Bay, which is the largest group of islands, using transects spaced 5 km apart and running perpendicular to the coast. All other islands in James and Hudson Bays were flown in a way to ensure complete coverage of the islands. The survey crew included one pilot and one data recorder in the front seats of the airplane and four active observers positioned in the rear of the airplane (two on the left and two on the right). We again erected an opaque barrier between the front and rear observers positioned in the rear of the airplane and conducted the survey identically to the mark-recapture protocol outlined above for the coastal zone of the high-density stratum. In this survey, the pilot and data recorder only indicated that they had detected a bear if it was directly on the flight line and thus unavailable to the observers in the rear of the aircraft.

Nearshore islands off Ontario coast

Along the coast of Ontario, there are a few small islands that are known to have large numbers of bears. Survey methods of distance sampling or mark-recapture are not well suited due to the small area of the islands and high bear density. Thus, these islands were surveyed separately using a total count methodology. They were comprehensively flown with the observer setup outlined above and bears were censused on them.

Québec coastline and nearshore islands

The survey of the Québec coastline and nearshore islands was similar to the 2012 survey (Obbard et al. 2015) and was limited to the coastline and nearshore islands. Considering the absence of polar bears observed during the 2016 survey within the 20 km inland portion of the survey (Obbard et al. 2018), consultations were conducted with the three Nunavik communities (Fig. 2) to review important areas where polar bears might be observed during late summer. All communities agreed that very few bears were present inland during that time of the year but one additional coastal area, south of Cape Jones down to the mouth of Seal River, was recommended to be surveyed and was added to the survey plan (MFFP, Unpublished). The Québec coastline and nearshore islands were surveyed using an A-Star 350 B2, from August 23rd to 27th. A single transect was flown along the coastline, flying at an altitude of approximately 150 m AGL at a ground speed of 150 km/h. All nearshore islands were surveyed in a way to ensure total coverage. The crew consisted of a pilot and navigator in the front of the helicopter and two rear observers positioned behind the pilot and navigator, with an opaque divider between the front and back in order to apply the double-observer mark-recapture methodology as described above for the surveying of the coastal zone of the high-density stratum in the Ontario mainland, coastline and Akimiski Island area.

Statistical analysis of Ontario mainland, coastline and Akimiski Island distance sampling surveys

A schematic outlining how each survey and area was analyzed is shown in Figure 8. The Ontario mainland, coastline and Akimiski Island distance sampling survey was analyzed using both (1) conventional distance sampling models with covariates (multiple covariate distance sampling [MCDS]; Marques and Buckland 2003, Marques and Buckland 2004), following the analysis of Obbard et al. (2018) as closely as possible to facilitate comparisons, and (2) mark-recapture distance sampling models (MRDS; Borchers et al. 1998, Laake and Borchers 2004) to allow modelling of imperfect detection on the transect line. MCDS models assume perfect detection of bears on the transect line and underestimate abundance if this assumption is violated (Buckland et al. 2001). MRDS models include a mark-recapture sub-model to estimate probability of detection on the line thereby avoiding the assumption of perfect detection anywhere (Borchers et al. 1998, Laake and Borchers 2004). Groups of bears, rather than individuals, were treated as the unit of observation. Estimates of group abundance were multiplied by the mean group size to convert to estimates of animal abundance. We conducted replicate MCDS and MRDS analyses including and excluding data from the coastal zone. Both types of models were implemented in the ‘mrds’ R package version 2.2.6 (Miller et al. 2019, Laake et al. 2022).

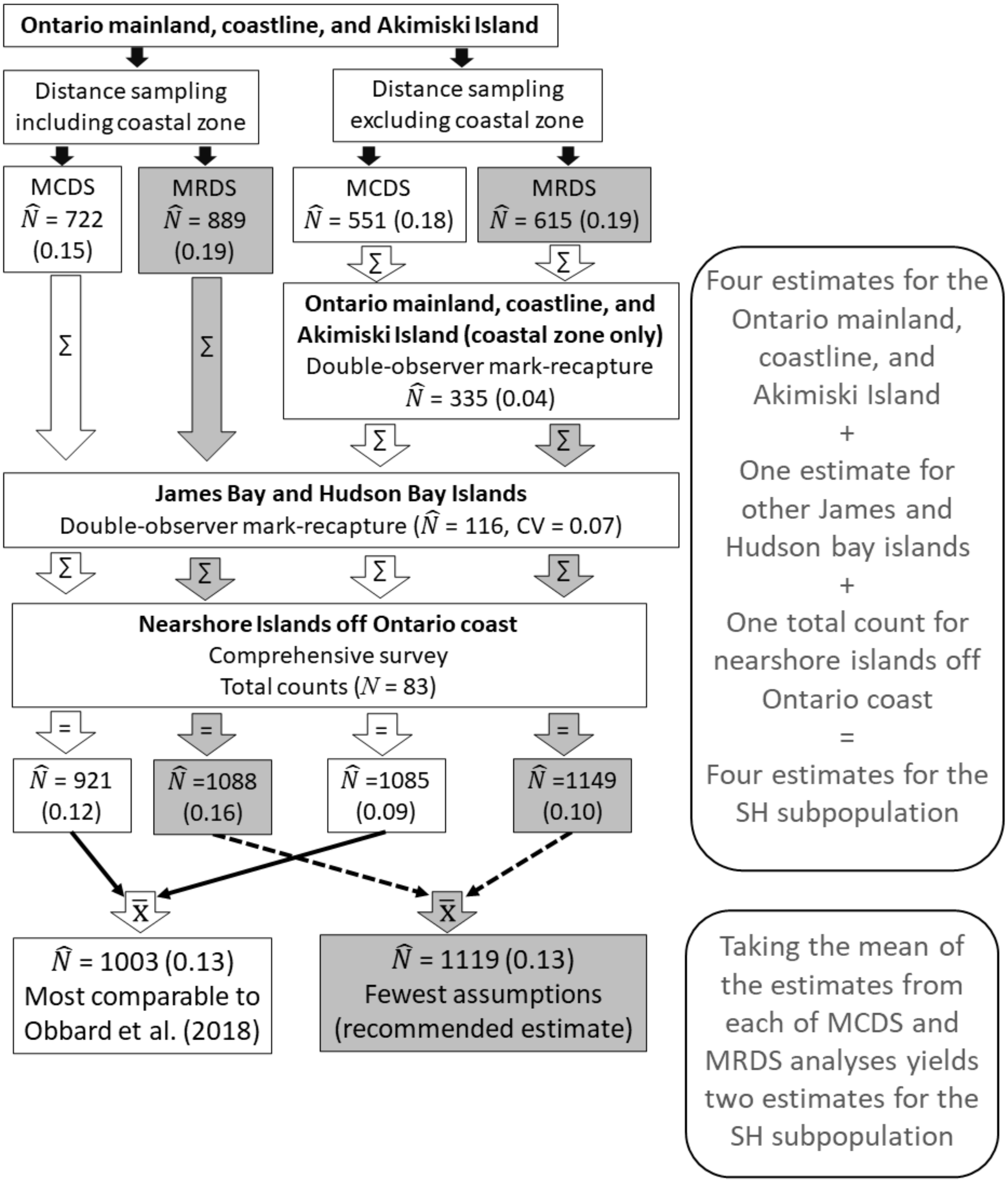


Figure 8: Schematic describing statistical analyses of data collected from different geographic areas and survey types. Geographic areas appear in bold and match those described under "survey design" above. ∑ indicates summation of estimates across different geographic areas, $\bar{x}$ indicates the mean across different estimates for the same geographic area. MCDS and MRDS refers to multiple covariate distance sampling and mark-recapture distance sampling analyses, respectively. Gray boxes and arrows indicate estimates derived using MRDS for the Ontario mainland, coastline and Akimiski Island area, while white arrows and boxes indicate estimates derived using MCDS for the Ontario mainland, coastline and Akimiski Island area. Note that because no bears were observed in the Québec coastline and nearshore islands portion of the study, that geographic region is not shown in the schematic.

For the MCDS analyses we right-truncated the data at 1750 m following Obbard et al. (2018) after verifying that distance sampling models fit the truncated data adequately (tests described below) and that abundances estimated from simple models were not sensitive to right-truncation distance. We initially considered unadjusted half-normal and hazard rate forms of the detection function as well as a uniform model with a cosine adjustment of order 1. Uniform models fit the data poorly or failed to converge so were not considered further. Potential covariates of the detection function included visibility, vegetation height, and vegetation density to match the analysis of Obbard et al. (2018). Covariates were evaluated using forward stepwise model selection where only covariates that reduced Akaike's Information Criterion (AIC; Burnham and Anderson 2002) were retained; vegetation height and density covariates were correlated so were not included in the same model. We checked whether adjustment terms (cosine of order 1 for the half-normal model, and simple polynomial of order 4 for the hazard rate model) improved the fit of the AIC-minimizing covariate models. We tested for significant ($\alpha = 0.05$) lack of fit using the X^2 goodness-of-fit test for binned distance data (Buckland et al. 2001, pp 69-71) and the distance sampling Cramér-von Mises test (Buckland et al. 2004, pp 388-389). The AIC-minimizing covariate model was selected for estimation (conditional on adequate fit), and final estimates were obtained by model averaging abundance estimates (as the AIC-weighted average abundance; Burnham and Anderson 2002) across hazard rate and half-normal models with the same covariate(s).

Data from the Ontario mainland, coastline and Akimiski Island distance sampling survey were also analyzed using MRDS models formulated for independent observers (Laake and Borchers 2004, Burt et al. 2014). Models with point independence rather than full independence were expected to be more appropriate for our data because the difference between front and rear observers' ability to see bears near the transects ensured that the correlation between detections from different observer positions increased with distance from the transect (Burt et al. 2014). We verified that simple point independence models reduced AIC relative to simple full independence models and used point independence models thereafter. We right-truncated at 2000 m because visibility was generally good in 2021 and exploratory analyses including goodness-of-fit testing indicated that this truncation distance provided a slightly better fit to simple DS models. We included distance as a covariate in all mark-recapture submodels (Buckland et al. 1993, Burt et al. 2014). We also considered a dummy covariate specific to the rear observers at short distances to account for their reduced probability of detecting groups of bears near the transect line (Wiig et al. 2022). The largest distance at which a group was recorded as unavailable to rear observers was 204 m, so all groups detected at this and shorter distances received a 1 for this "blind-spot" covariate. Other potential covariates of the MR submodel were group size, observer position (front or rear), side, the interaction between position and side, visibility, vegetation height, vegetation density, glare, and stratum (high or low density). Because vegetation height and density were correlated but describe potentially different effects of vegetation on observers' ability to detect bears, we also evaluated a combined vegetation covariate (Table 1); only one vegetation covariate was included in any submodel. Potential covariates of the distance sampling submodel included group size, side, visibility, vegetation height, vegetation density, the combined vegetation height and density covariate, glare, and stratum. After exploratory analyses we excluded the "activity" covariate because estimated effects were weak and indicated that stationary bears were more likely to be detected, including at longer distances, than moving bears.

Table 1. Definition of vegetation covariate representing the combination of vegetation height and density. The vegetation height covariate was recorded in the field on a 3-point scale with a height of 1 indicating vegetation was <1 m, 2 indicating 1-3m and 3 indicating >3 m. The vegetation density covariate was recorded in the field on a 3-point scale with a density of 1 indicating sparse vegetation, 2, indicating moderate and 3 dense.

Vegetation height	Vegetation density	Combined vegetation covariate
1 or 2	1	1
1 or 2	2	2
1 or 2	3	not present in data
3	1	2
3	2	3
3	3	4

We evaluated support for forms of the detection function (unadjusted half-normal or hazard rate) and covariates using a forward stepwise model selection procedure intended to avoid overfitting and the inclusion of uninformative covariates in estimating models. Covariates that increased AIC relative to a simpler model without that covariate were excluded, covariates that reduced AIC were retained but if the reduction was < 2.0 we also considered parameter-reduced models excluding those covariates. This approach differed slightly from the above analysis because here we considered more covariates and thus needed to evaluate more combinations of covariates. Thus, we required a larger reduction in AIC to avoid evaluating a cumbersome number of models. An exception to this procedure was that, following Northrup and Howe (2019), we considered a model with main effects of side and position and their two-way interaction in all mark-recapture submodels even if side and position were not supported as main effects alone. We conducted model selection in 3 steps. First, we held the distance sampling model constant as the unadjusted half-normal model with no covariates and evaluated covariates of the mark-recapture model. Next, we evaluated forms and covariates of the distance sampling model while holding the mark-recapture model constant at the AIC-minimizing model. Lastly, we created a set of models that was comprised of all combinations of the supported (ΔAIC < 2) mark-recapture and distance sampling submodels. We checked whether the adjustment terms described above for MCDS models improved the fit of the AIC-minimizing distance sampling submodels. Before estimating abundance we checked for significant ($\alpha = 0.05$) lack of fit using X^2 tests across distance intervals for both the mark-recapture and distance sampling submodels, the total X^2 value across submodels, and the Cramér-von Mises test. Final MRDS estimates of abundance were obtained by model averaging across models with supported covariates and parameter-reduced models in the case of weakly-supported (ΔAIC < 2) covariates.

In both the MCDS and MRDS analyses, the variance of the abundance of individual bears combined three components of variance using the delta method (Buckland et al. 2001, Miller et al. 2019): the empirical variance of the encounter rate among transects (here estimated using Fewster et al. 's [2009] estimator "S2" for systematic designs), the variance of detection probability obtained from the fitted model estimated using standard maximum likelihood methods, and the variance of group size. Where estimates were calculated by model averaging, model selection uncertainty also contributed to the variance of bear abundance (Burnham and Anderson 2002).

We post-stratified estimates of abundance by age-sex category (adult females, adult males, subadults, yearlings, and cubs) to obtain age-sex class specific estimates of abundance in the Ontario mainland, coastline and Akimiski Island portion of the survey. This was achieved by combining the estimated probability of detecting clusters of bears (and its variance) from the AIC-minimizing model fit to data from all clusters with age-sex class specific group sizes.

Statistical analysis of double-observer mark-recapture surveys

The Ontario mainland, coastline and Akimiski Island coastal zone mark-recapture helicopter survey and the James Bay and Hudson Bay islands fixed-wing mark-recapture surveys were analyzed using mark-recapture models for closed populations (Huggins 1989) implemented in the ‘RMark’ R package version 2.2.7 (Laake 2013, Laake et al. 2019). We conducted separate analyses of data obtained from the helicopter survey and the combined fixed-wing surveys (Fig. 8). Potential covariates of detection probability included observer position (front or rear, modelled as distinct temporal sampling occasions), group size, visibility, vegetation height, vegetation density, and position of the group relative to the aircraft (left, right, or under, coded as “under” where the group was recorded as unavailable to the rear observer). We fixed detection probability by the rear observers to 0 for groups that passed “under” the aircraft for both the fixed-wing and helicopter surveys. We evaluated support for covariates using the same forward stepwise procedure described above for the mark-recapture and distance sampling submodels of MRDS models, except that we used the small sample bias-corrected version of AIC (AIC*c*; Burnham and Anderson 2002) rather than AIC. We obtained final estimates of the number of groups of bears and its unconditional variance by model-averaging abundance estimates across models with supported covariates, and parameter-reduced models in cases of weakly supported (ΔAIC < 2) covariates. We estimated the number of individual bears by multiplying by mean group size and included the variance of group size in the variance of the number of bears using the delta method.

We did not detect any bears during the mainland Québec coastal and nearshore island survey. As such, no statistical analyses were applied.

Total abundance estimates

The above analyses produced four separate estimates of bear abundance in the Ontario mainland, coastline and Akimiski Island area (see also Fig. 8): 1) an MCDS estimate for the entirety of the area (i.e., the areas overlain by the green, orange and purple polygons in Fig. 6), 2) an MCDS estimate for the low-density stratum and the inland zone of the high-density stratum (i.e., excluding the coastal zone, so the orange and purple polygons in Fig. 6) plus the estimate of the number of bears in the coastal zone (the area in green in Fig. 6) from the double-observer mark-recapture analysis, 3) an MRDS estimate for the entirety of the area (i.e., the areas overlain by the green, orange and purple polygons in Fig. 6), and 4) an MRDS estimate for the low-density stratum and the inland zone of the high-density stratum (i.e., excluding the coastal zone, so the orange and purple polygons in Fig. 6) plus the estimate of the number of bears in the coastal zone (the area in green in Fig. 6) from the mark-recapture analysis. We added the estimated number of bears on the James Bay and Hudson Bay Islands, and the census number of bears on small nearshore islands off the Ontario coast, to each of the four final estimates for the Ontario mainland, coastline and Akimiski Island area to generate estimates for the SH subpopulation. Finally, we produced two final estimates of the SH subpopulation as the mean of two subpopulation-level estimates: those calculated from estimates 1 and 2 above for the Ontario mainland, coastline and Akimiski Island area, and those calculated from estimates 3 and 4 above (see Fig. 8). Unconditional variances around these estimates were calculated in a model averaging framework assigning the two estimates equal weight. We present log-normal confidence intervals around all estimates of bear abundance. All analyses were performed using R software version 4.2.0 (R Core Development Team 2022).

RESULTS

Ontario mainland, coastline and Akimiski Island area

We detected 138 groups of bears on distance sampling transects on the Ontario mainland, coastline and Akimiski Island area, 88 excluding the coastal zone. Right-truncating at 1750 m for the MCDS analysis removed 9% of observations from both data sets, leaving 125 and 80 groups in data including and excluding the coastal zone, respectively. Right-truncating at 2000 m for the MRDS analysis removed 8% of observations from the complete data set and 7% of observations from data excluding the coastal zone, leaving 127 and 82 groups in data including and excluding the coastal zone, respectively.

In the MCDS analysis of the dataset including the coastal zone, the half-normal model without covariates minimized AIC. However, half-normal and hazard rate models with the vegetation density covariate had similar support with ΔAIC of 0.61 and 0.76, respectively (Table S1), so, for the sake of consistency with Obbard et al. (2018), we estimated abundance by model averaging across these two models (Table 2). Visibility was the only supported covariate in data excluding the coastal zone; half-normal and hazard rate models with this covariate had similar support, and all other models had ΔAIC > 2 (Table S2), so we estimated abundance by model averaging across these two models (Table 2). All MCDS models considered for estimation provided adequate fits to the data (*P*-values associated with the X^2 test for binned distance data and the Cramér-von Mises tests were all > 0.30). Adjustment terms did not improve fit to either data set.

Table 2. Abundance estimates ($\widehat{N}$), standard errors (SE), coefficients of variation (CV) and 95% confidence intervals from multiple covariate distance sampling (MCDS) and mark-recapture distance sampling (MRDS) analyses of polar bear data including or excluding the coastal zone of the high-density stratum for the Ontario mainland, coastline and Akimiski island area only.

Analysis type	Coastal zone	$\widehat{N}$	SE	CV	95% CI
MCDS	Included	722	111	0.15	535 – 974
MCDS	Excluded	551	99	0.18	388 – 781
MRDS	Included	889	170	0.19	613 – 1288
MRDS	Excluded	615	119	0.19	422 – 897

In the MRDS analysis of the complete data set, the blind-spot covariate, observer position, side, and glare were supported covariates of the mark-recapture submodel and the interaction between position and side and visibility were weakly supported (ΔAIC < 2 relative to simpler models) so additional models including and excluding these latter covariates were considered. Three submodels with all supported covariates and different combinations of weakly supported covariates had ΔAIC < 2 and were crossed with supported distance sampling submodels. Glare was supported as a covariate of the distance sampling submodel (Fig. 9). The combined vegetation covariate was also supported (Fig. 9), but ΔAIC was < 2 in the case of half-normal models so we considered models excluding it. Adjustment terms did not improve fit. Three submodels had ΔAIC < 2 and were crossed with the three supported mark-recapture models. All nine supported MRDS models (Table S3) fit the data adequately (*P*-values associated with the total X^2 value across distance sampling and mark-recapture submodels and the Cramér-von Mises tests were all > 0.65) and were included in model-averaged estimates of abundance (Table 2).

When data from the coastal zone were excluded, the blind spot covariate, observer position, side, the interaction between position and side, visibility, and glare were supported covariates of the mark-recapture submodel in the MRDS analysis. However, models with the visibility or glare covariates exhibited lack of fit that was sometimes significant at $\alpha = 0.05$ and always significant at $\alpha = 0.10$ (*P*-values associated with the total X^2 value ranged from 0.03 – 0.08); furthermore, these models yielded unrealistically high estimates of abundance, suggesting data were insufficient to support this level of model complexity. We therefore combined only the mark-recapture submodel with the blind spot covariate, position, side, and the interaction between position and side with supported distance sampling submodels. All other submodels that fit well and yielded reasonable abundance estimates had ΔAIC > 2 relative to this submodel. Only visibility was supported as a covariate of the distance sampling submodel; it reduced AIC of the hazard rate model by < 2 so we retained models excluding it and combined four distance sampling submodels (half-normal and hazard rate with and without the visibility covariate) with the selected mark-recapture submodel (Table S4). Adjustment terms did not improve fit. All four of these models fit the data adequately and were included in model averaged estimates of abundance (Table 2). MCDS and MRDS estimates of abundance were sensitive to the form of the detection function (half-normal or hazard rate) and less sensitive to covariates.

Post-stratification of the Ontario mainland, coastline and Akimiski island observations by age-sex class based on the distance sampling data, including the coastal zone, suggests an adult sex ratio strongly skewed towards females (Table 3). Raw observations from the coastal mark-recapture survey showed a strongly male biased sex ratio (Table 4).

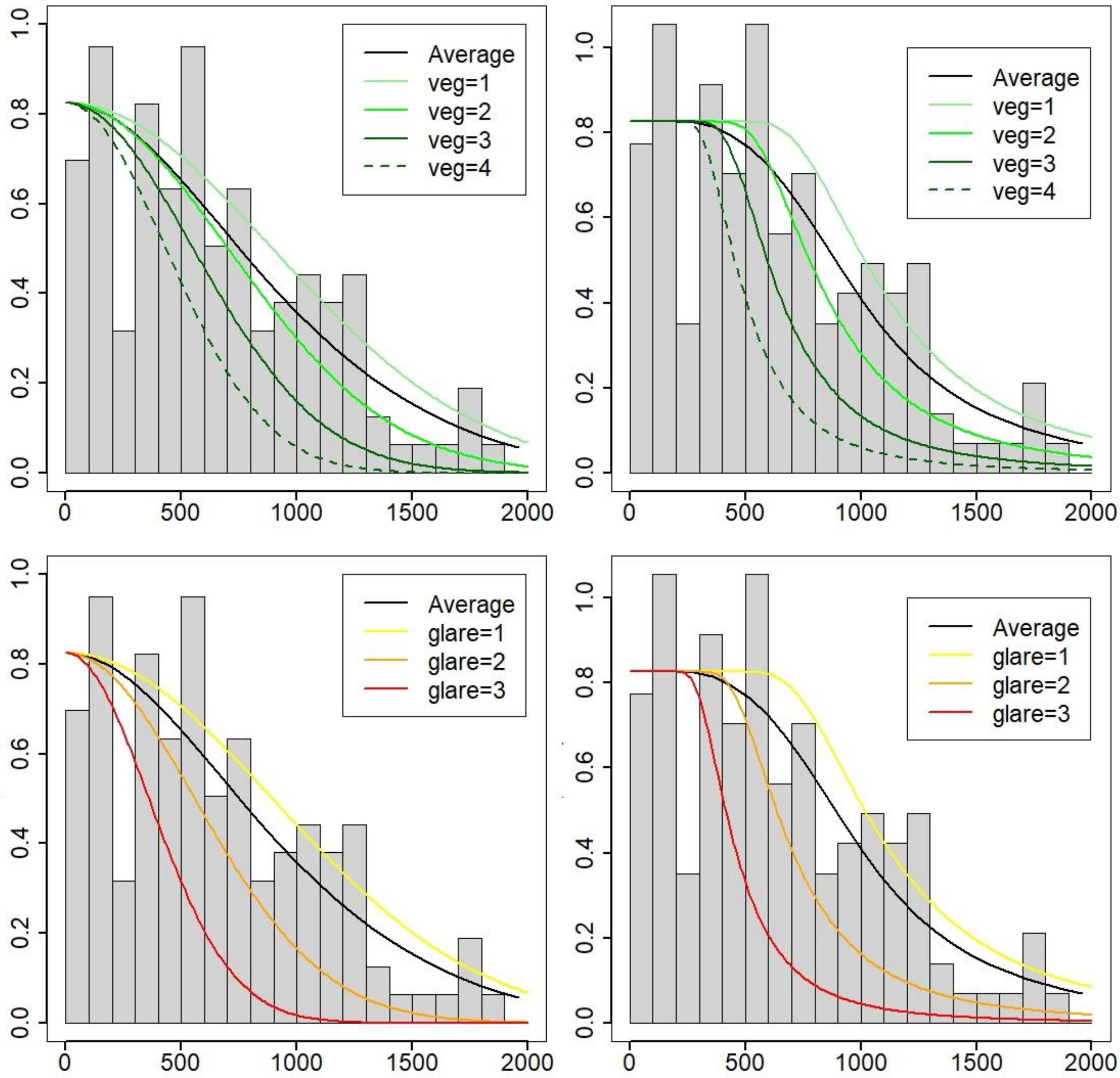


Figure 9. Half-normal (left column) and hazard rate (right column) detection functions estimated from the top two AIC-ranked mark-recapture distance sampling models fit to complete data from SH polar bears sighted from distance sampling transects in 2021 in the Ontario mainland, coastline and Akimiski island area, showing effects of supported covariates on the scale of the detection functions (the combined vegetation covariate and glare). Both models included the same covariates of both submodels; only key functions differed. The half-normal model ranked 1st and the hazard rate model had ΔAIC = 1.3. Top row shows the effect of the vegetation, bottom row shows the effect of glare. When plotting effects of one covariate, the other covariate was held constant at the mean value in the data. X-axes show distance from the transect in meters, y-axes show probability of detection.

Table 3. Estimates of abundance ($\widehat{N}$), standard errors (SE), coefficients of variation (CV), lower 95% confidence limit (LCL), upper 95% confidence limit (UCL) and the mean proportion (Prop.) of the total estimate comprised of that sex and age class, obtained from post-stratification of MRDS model fit to distance sampling observations of polar bears in the Ontario mainland, coastline and Akimiski Island portion of the survey (including coastal zone) in 2021.

Age-sex class	$\widehat{N}$	SE	CV	LCL	UCL	Prop.
Adult female	366	70	0.19	251	533	0.40
Adult male	173	71	0.41	79	378	0.19
Subadult	59	21	0.36	30	118	0.06
Yearling	156	38	0.24	98	250	0.17
COY	167	52	0.31	91	305	0.18

Table 4. Proportions of polar bears of different sex and age classes observed during distance sampling surveys of the entire Ontario mainland, coastline, and Akimiski Island portion of SH (including the coastal zone) and from mark-recapture surveys of only the coastal zone within that portion of the study area, 2011 – 2021.

	Year	Adult female	Adult male	Subadult	Yearling	COY
Distance sampling						
	2011	0.36	0.20	0.08	0.15	0.19
	2016	0.34	0.19	0.06	0.05	0.30
	2021	0.37	0.20	0.07	0.18	0.18
Coastal mark-recapture						
	2011	0.20	0.40	0.13	0.12	0.15
	2016	0.19	0.52	0.08	0.03	0.17
	2018	0.19	0.55	0.09	0.07	0.10
	2021	0.22	0.43	0.11	0.12	0.10

No covariates of detection probability were supported in mark-recapture analyses of data from the helicopter survey of the coastal zone. Probabilities of detection were high (observer specific probability of detection = 0.87 from the null model, equating to approximately 0.98 probability that either observer detected a group of bears) and estimates of abundance were similar across all models. Multiplying the estimated number of groups from the null model by mean group size (1.567; SE 0.063) yielded an estimate of 335 bears (SE 13.9, CV 0.04, 95% CI = 309 – 363).

James Bay and Hudson Bay islands (excluding Akimiski Island)

Side and group size were weakly supported covariates in the mark-recapture analysis of data from the fixed wing survey of the James and Hudson Bay Islands. Estimated probabilities of detection were again high (observer specific probability of detection =

0.841 from the null model, equating to approximately 0.97 probability that either observer detected a group of bears) and estimates of abundance were similar across models. The estimated number of groups obtained by model averaging multiplied by mean group size (1.455; SE 0.090) yielded an estimate of 116 bears (SE 7.93, CV 0.07, 95 % CI = 102 – 133).

Nearshore islands off the Ontario coast

The comprehensive survey of the nearshore islands off the Ontario coast yielded a total count of eighty-three (83) bears.

Québec coastline and nearshore islands

No bears were observed along the Québec mainland coast and nearshore islands.

Total abundance estimates

Estimates of total abundance at the subpopulation level ranged from 921 to 1149 and were lower when using the MCDS approach which assumed perfect detection on the transect line during distance sampling surveys (Table 5). In total, we saw 148 family groups during the whole survey, including those seen while off transect or transiting. Seventy-five of these were females with cubs of the year and 73 with yearlings. The average cub of the year litter size was 1.57 ($\sigma = 0.52$) and the average yearling litter size was 1.47 ($\sigma = 0.50$).

Table 5. Estimates of subpopulation-wide abundance ($\widehat{N}$), standard errors (SE), coefficients of variation (CV), lower 95% confidence limit (LCL) and upper 95% confidence limit (UCL) for polar bears in the Southern Hudson Bay subpopulation. Six estimates are presented representing either multiple covariate distance sampling (MCDS) or mark-recapture distance sampling (MRDS), excluding the coastal zone, including the coastal zone or averaging across these two approaches. All 6 abundance estimates include the 116 bears estimated to be on the James Bay and Hudson Bay islands as well as the 83 bears counted on the nearshore islands off the Ontario coast.

Estimate	Method and areas included	$\widehat{N}$	SE	CV	LCL	UCL
1	MCDS including coastal zone	921	111	0.121	727	1166
2	MCDS excluding coastal + coastal zone MR	1085	100	0.092	905	1300
3	Mean of 1 & 2	1003	134	0.134	773	1302
4	MRDS including coastal zone	1088	170	0.156	802	1474
5	MRDS excluding coastal + coastal zone MR	1149	120	0.105	937	1410
6	Mean of 4 & 5	1119	150	0.134	860	1454

Discussion

The number of polar bears present in the SH subpopulation at the time of the 2021 survey was substantially higher compared to the last comprehensive survey conducted in 2016. In 2016, the subpopulation estimate was 780 (95% confidence interval 590-1029; Obbard et al. 2018), which represented a 17% decline from 2011/12 when the subpopulation was estimated at 943 (95% confidence interval 658-1350; Obbard et al. 2015). In our current work, we produced two separate estimates, one (N = 1003 95% CI = 773-1302) that assumed perfect detection on the transect line as Obbard et al. (2018) did to allow for direct comparison and one (N = 1119 95% CI 860-1454) that took advantage of a novel approach to estimating the probability of detection on the transect line while accounting for the blind spot affecting rear observers (Wiig et al. 2022). The former estimate is most comparable to the 2016 estimate, but the latter is a more robust estimate of the true subpopulation size in 2021. Both estimates indicate a greater number of bears within this subpopulation than in 2016, with the former estimate suggesting a 29% increase in the number of bears found within the subpopulation in 2021 compared to 2016.

The greater number of bears in SH in 2021 compared to 2016 has two plausible biological drivers based on the results of this survey and other available lines of evidence, both of which may be at play to varying degrees: 1) annual variation in the on-land distribution of bears in SH and WH, and 2) an increase in population growth rate due to reduced mortality, increased birth rate or both. At the writing of this report, we do not have definitive evidence for either driver, but discuss the existing evidence for each of these in turn. First, it seems likely that there was some movement of bears into SH from the adjacent WH subpopulation in 2021. An increase of nearly 30% in 5 years seems highly implausible for a species such as polar bears that has a slow life history strategy. Further, the 2016 survey showed very few yearlings, and a survey of only the coastal area in 2018 found even fewer bears than in 2016 in this portion of the subpopulation. These findings suggest that an even greater rate of increase would have to have occurred between 2018 and 2021, making it highly unlikely that all of the increase from 2016 to 2021 was from greater reproductive output or reduced mortality alone. A simultaneous survey of WH (Atkinson et al. 2022) indicated a decline of 224 bears in WH from 2016 to 2021, which numerically is the same as the increase in the estimate of SH abundance from Obbard et al. (2018) and our 2021 survey. Further, genetic identification of individuals sampled through biopsy darting conducted along the coast of SH and WH indicated that > 20% of the bears sampled in SH in 2021 had previously been sampled exclusively in WH (McGeachy et al. 2023). These joint lines of evidence suggest that there is variation in the annual on-land distribution of bears between SH and WH, with more of these bears in SH in 2021. Although the boundary between WH and SH, in northwestern Ontario, was based in part on movement and mark-recapture data, there is no major physiographic feature present and there are large aggregations of bears on offshore islands and peninsulas near the boundary. Thus, minor variation in the distribution of these bears could greatly shift the number of individuals present in WH or SH. Prevett and Kolenosky (1982) suggested that movements of large numbers of bears occurred between the southern Manitoba coast of Hudson Bay and Ontario, though this finding was not corroborated by Stirling et al. (2004) using surveys conducted earlier in the ice-free season. Derocher and Stirling (1990), focusing on the area of WH directly south of Churchill, MB likewise did not document movements between the two subpopulations, but did not cover the area of WH closest to SH where relatively minor annual variation in distribution could lead to large shifts in the number of bears present in each subpopulation. Further, collaring data from female bears shows generally high fidelity to onshore areas (Stirling et al. 2004, Obbard and Middel 2012). However, more recently, Cherry et al. (2013) showed that ice conditions were an important predictor of annual fidelity to onshore areas in WH. Specifically, they found that when there was greater ice later in the season in SH relative to WH, bears collared in WH tended to come ashore further from their collaring location. Further, they predicted greater declines in seasonal fidelity to onshore areas with continued sea-ice decline. The biopsy darting work (McGeachy et al. 2023), in combination with ongoing physical capture (ECCC, unpublished data) covered the coast of WH from the border between Manitoba and Nunavut to the WH-SH border, along with much of the SH coast and is the most comprehensive data available to date on individual movements; these data are more comprehensive in coverage than either Derocher and Stirling (1990) or Prevett and Kolenosky (1982) and use more effective methods for documenting annual movement of individuals of all sex and ages classes than does telemetry or aerial surveys (e.g., Stirling et al. 2004, Obbard and Middel 2012).

In contrast to the above evidence for annual variation in distribution of bears leading to the increase in SH, it is possible that this increase was influenced in part by improved demographic rates in SH. Several lines of evidence support that the decline in WH from 2016 to 2021 was at least partially driven by reduced reproduction. If this is the case, then the increase in SH could not be solely driven by distribution shift. First, reproduction and recruitment in WH appear to have been low throughout the last decade relative to SH and other polar bear subpopulations (Atkinson et al. 2022). Specifically, cubs of the year comprised 7%, 11% and 9% of observations of bears in 2011, 2016 and 2021 in WH, while yearlings comprised 3%, 3% and 9% (Stapleton et al. 2014, Dyck et al. 2017, Atkinson et al. 2022). In comparison, cubs of the year comprised 16%, 19% and 14% of observations in SH in 2011, 2016 and 2021 and yearlings comprised 12%, 5% and 13% of observations (Obbard et al. 2015, Obbard et al. 2018). Further, physical mark-recapture in part of WH indicates there have been few yearlings during many of the last 10 years (ECCC unpublished data). These numbers alone suggest reproduction is substantially greater in SH than WH. WH also has seen strong evidence of changes in sex and age class ratios across the three surveys, with declines in adult females and sub-adults (Atkinson et al. 2022). Observations data from the 2011, 2016 and 2021 surveys in SH however show quite consistent sex and age structure. However, the number of yearlings in 2021 was high and indicates a rebound from the particularly low numbers seen in 2016 (Obbard et al. 2018). Annual variability in survival of COYs to yearlings is not surprising as autumn yearling litter sizes are highly variable (Derocher and Stirling 1995). We also note that the two years preceding 2021 were two of the three years with the longest duration of sea-ice since 2011 (Figs. 10 & 11). These conditions would have been favorable for high reproductive output and survival of cubs in the previous two years. Importantly, with continued warming, these conditions are unlikely to persist and we expect low recruitment in the coming years.

The above numbers suggest that in recent years, demography is different in WH and SH, with what appears to be lower reproduction and recruitment in WH. If this is the case, then the decline seen in WH by Atkinson et al. (2022) may not be all attributable to distribution shifts of bears to SH. Following, the increase in SH would have to be at least partially due to increased population growth rate. This potential is supported by the fact that ice conditions have generally been good over the last 5 years relative to the time period between 2011 and 2016 (Fig. 10) and that SH appears to have a high capacity for growth (Regehr et al. 2021). Further, polar bear harvest in SH was lower between 2016 and 2021 than between 2010 and 2015 (37.8 bears per year compared to 58.8 bears per year; https://www.polarbearscanada.ca/en/polar-bears-canada/canadas-polar-bear-subpopulations; accessed July 22, 2022). This decrease was in part driven by the exceptionally large harvest of 104 bears in the 2010/2011 harvest season, of which many were female. Such a large increase in annual harvest must have had downstream negative demographic effects due to the increased harvest of adult females, subsequently potentially depressing growth for a few years. Thus, it seems plausible that the high harvest in 2010/11 and higher average harvest early in the last decade, along with relatively poor ice years, could have driven a decline between 2011 and 2016. In contrast, a subsequent rebound to 2021 levels could be due to lower annual harvests with the resulting downstream positive demographic effects combined with better ice conditions that resulted in higher juvenile survival. However, we note again that a 29% increase over 5 years is highly unlikely for polar bears without distribution shift playing some role. Lastly, it is possible that the apparent increase in SH between 2016 and 2021 was simply sampling variance in one or both years, whereby the true difference in numbers between the surveys was exaggerated. We note that it is equally likely that the difference was underestimated, however.

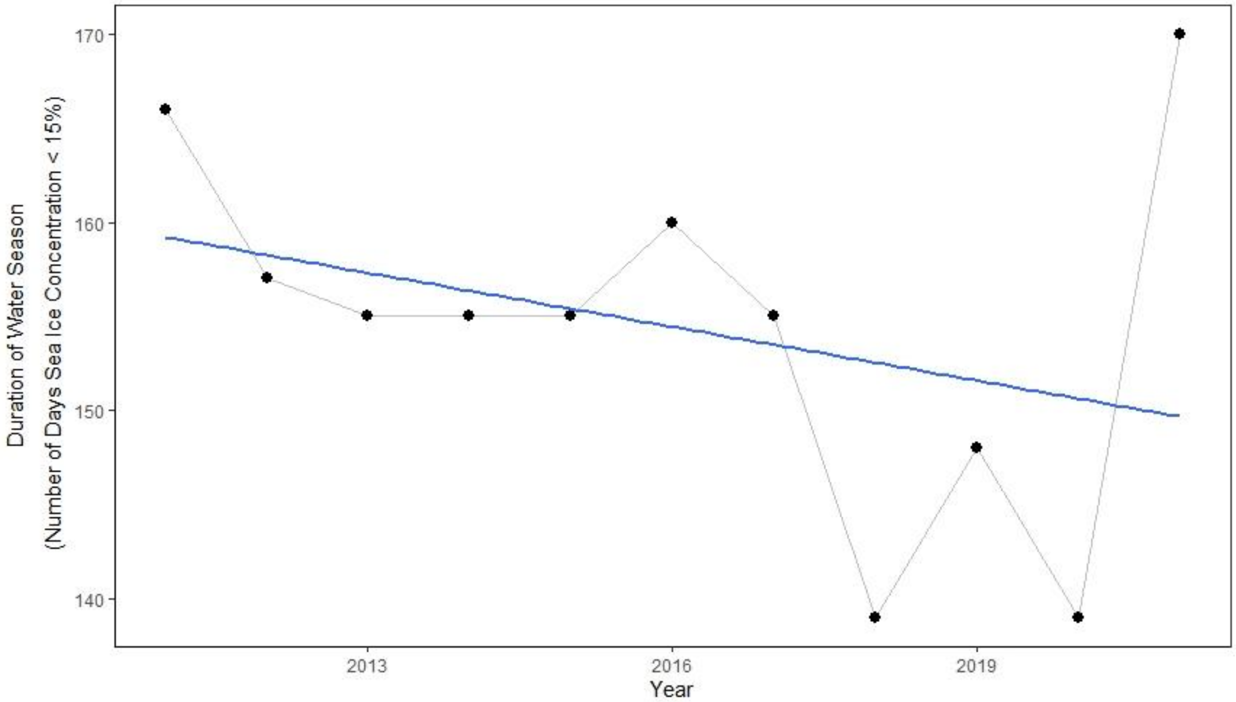
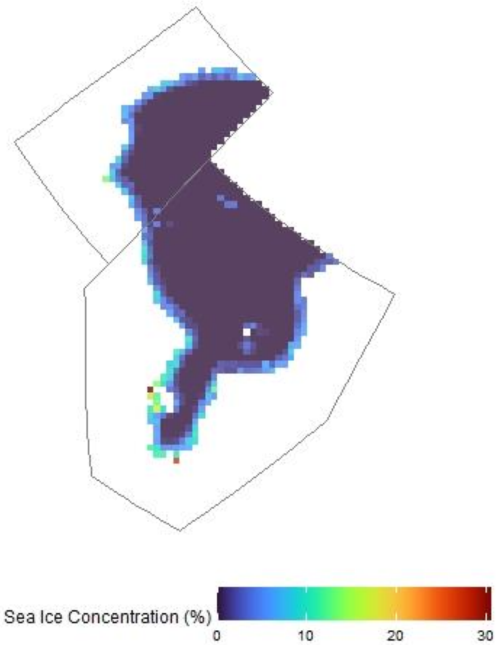
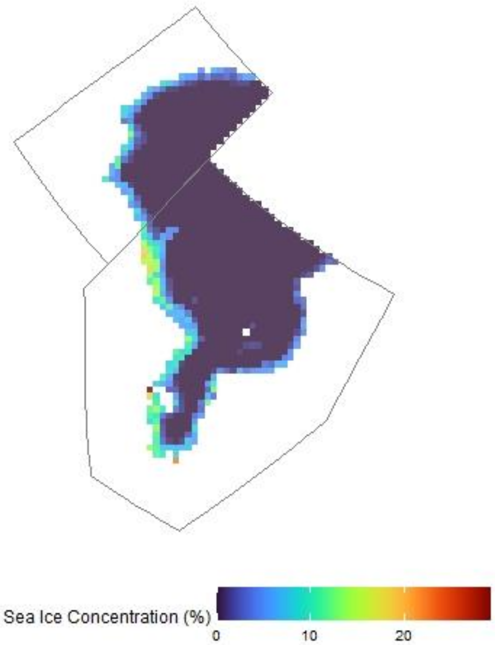


Figure 10. Duration of ice-free season in the combined Western and Southern Hudson Bay polar bear subpopulations, calculated as the number of days in which the combined area had less than 15% sea-ice concentration. The blue line represents a trend fit to the ice-free days.

July 15 - August 15, 2011



July 15 - August 15, 2012



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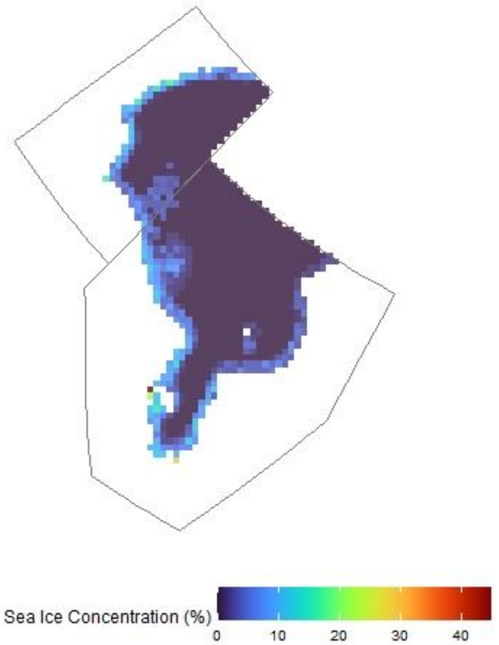
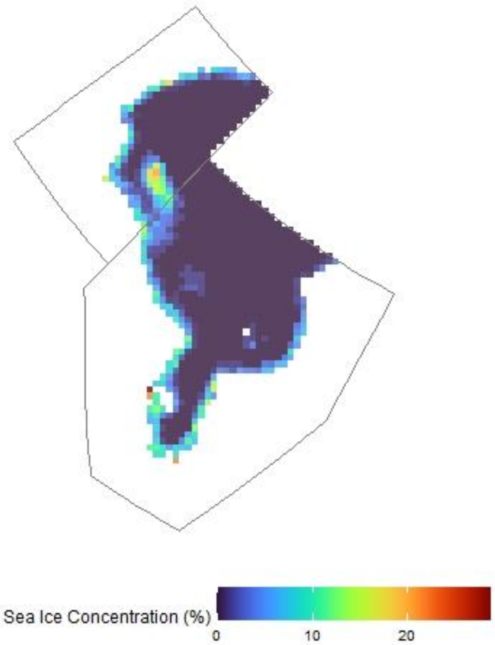
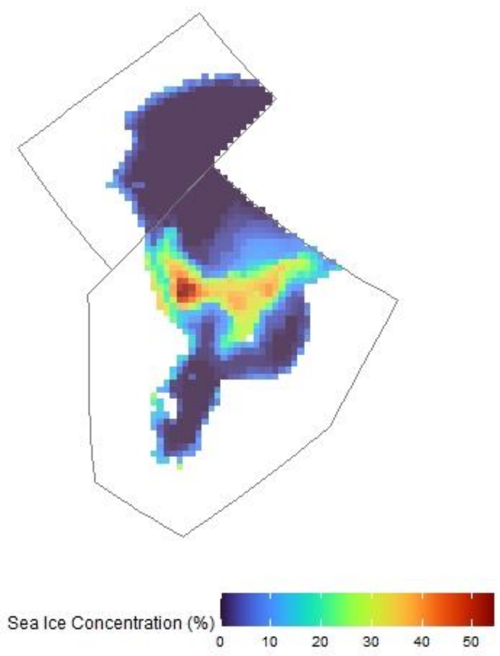
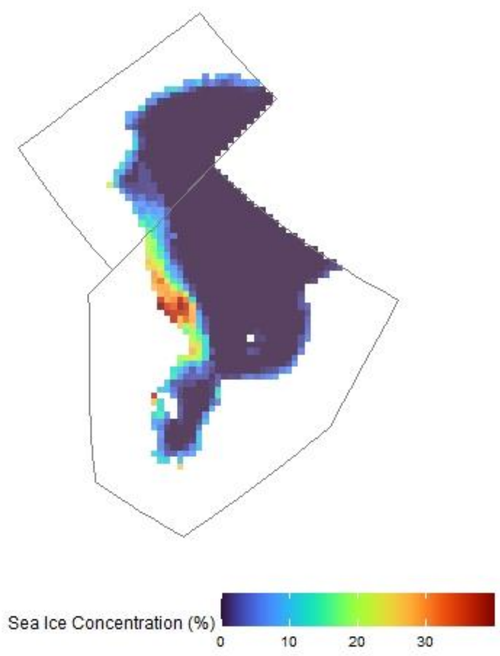
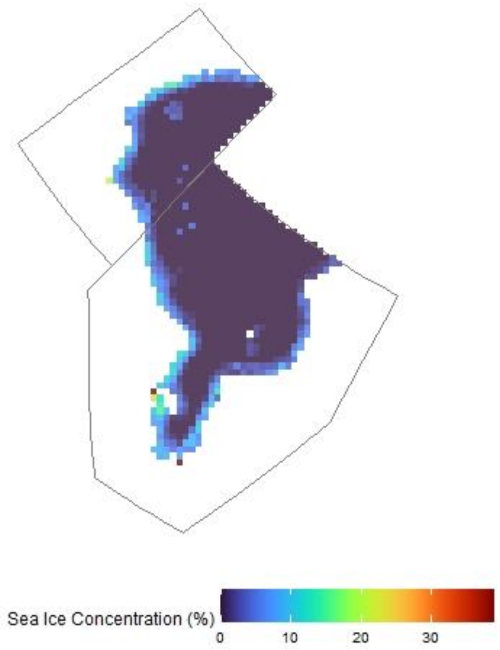


July 15 - August 15, 2014




July 15 - August 15, 2015


July 15 - August 15, 2016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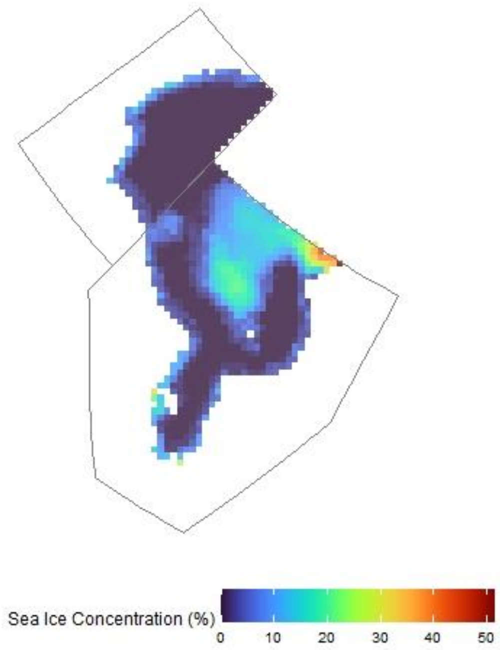

July 15 - August 15, 2018

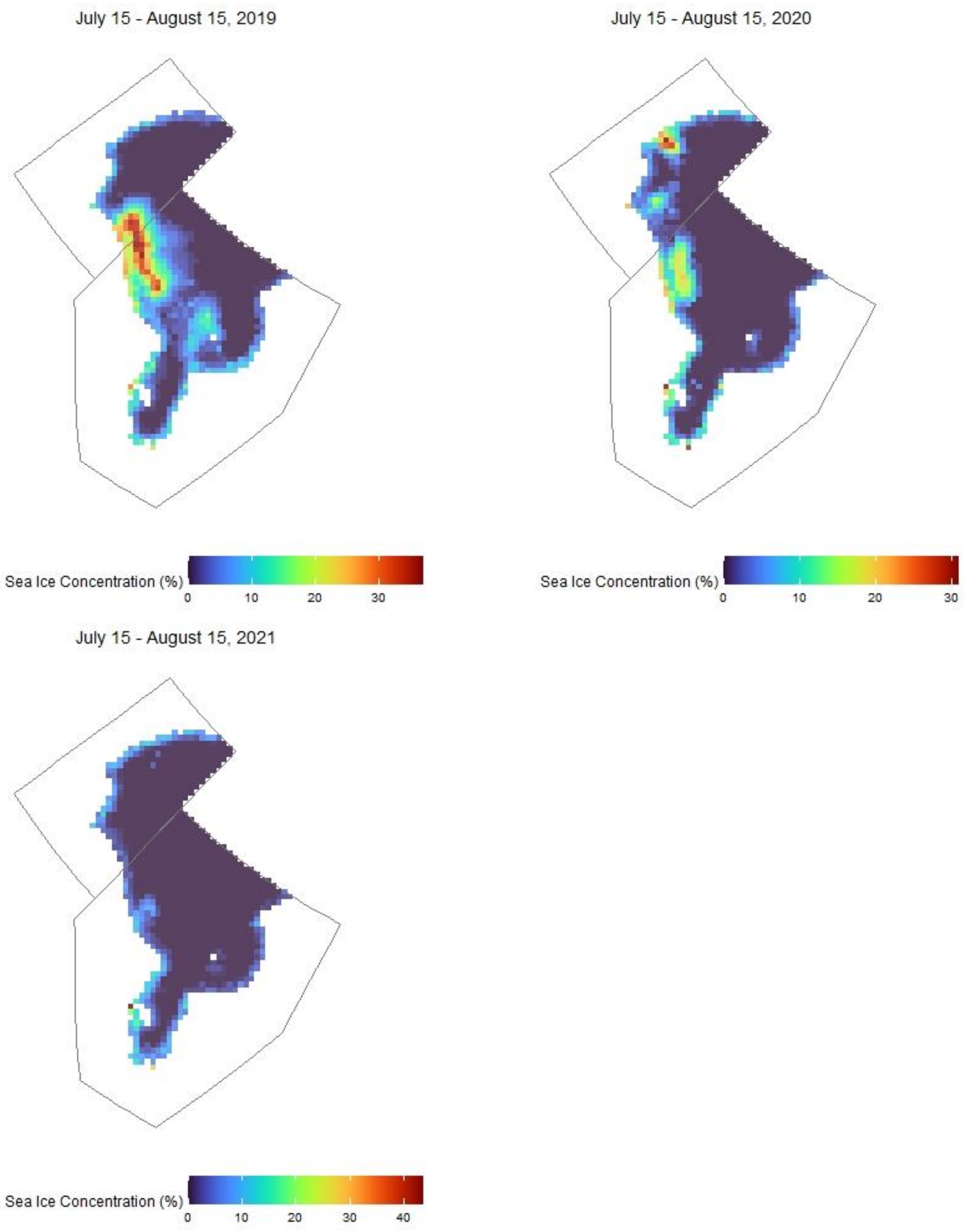
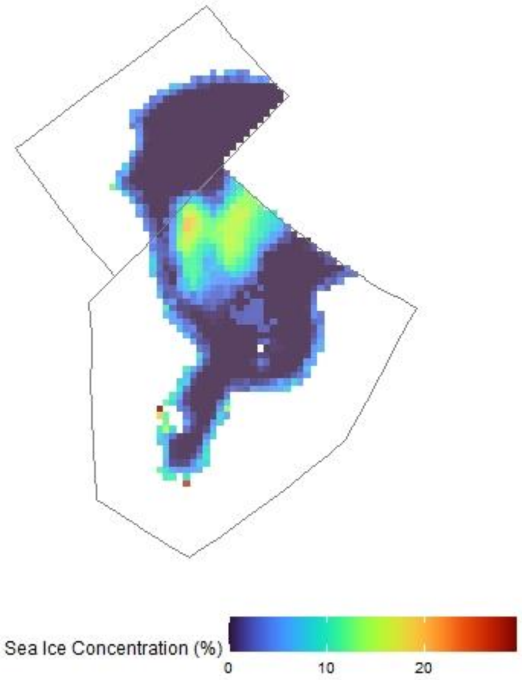
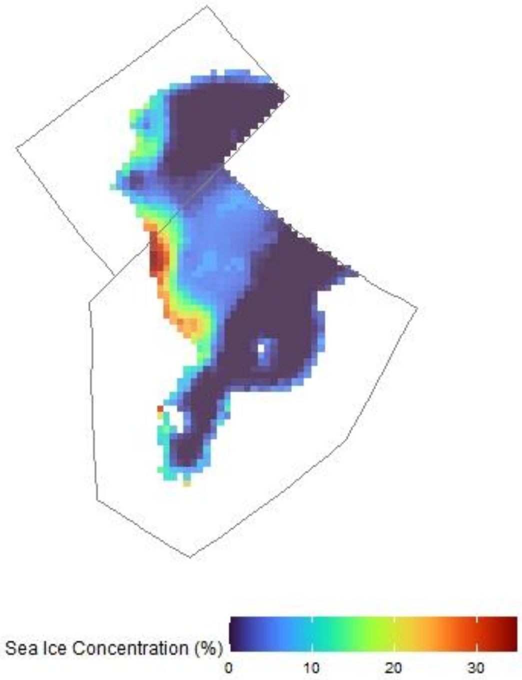


Figure 11. Average sea-ice concentration from July 15 through August 15 for each year from 2011 through 2021 for the Western and Southern Hudson Bay polar bear subpopulations.

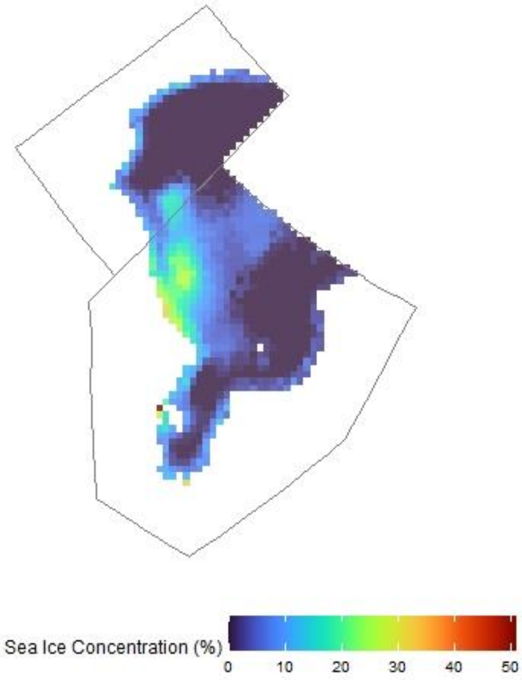
July 2011



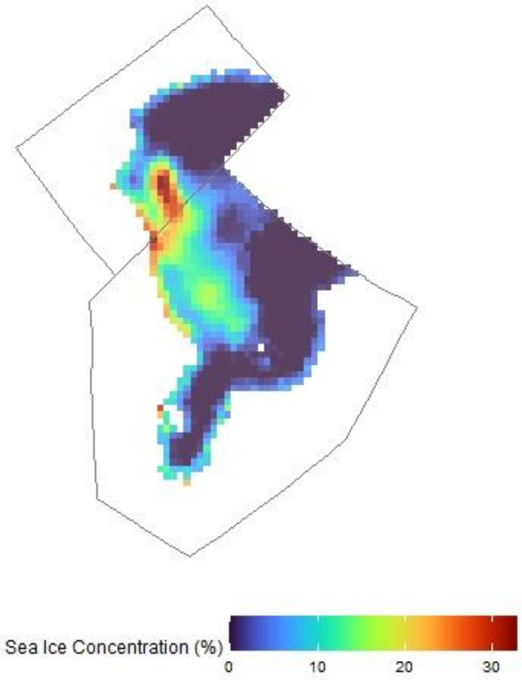
July 2012

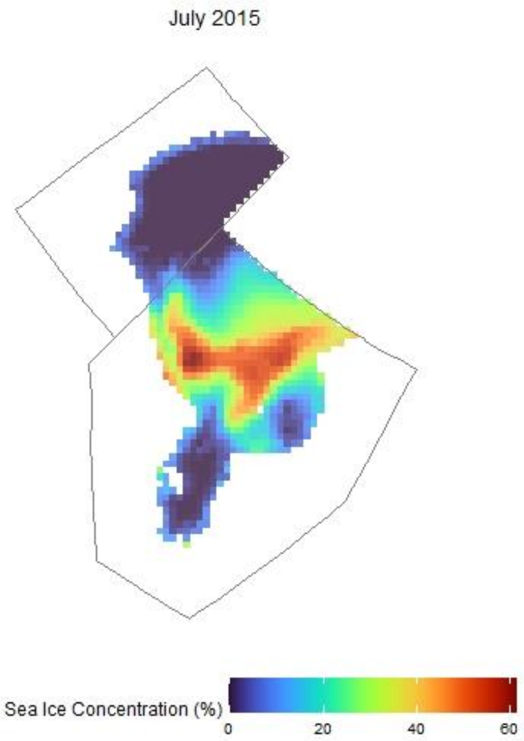


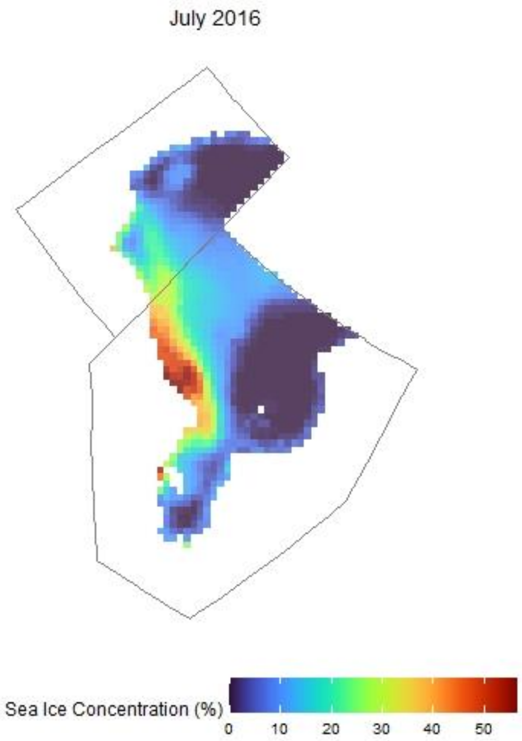
July 2013

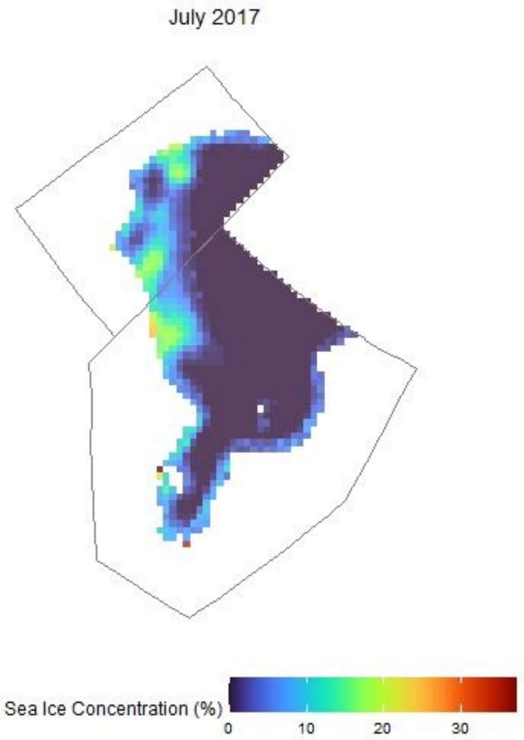


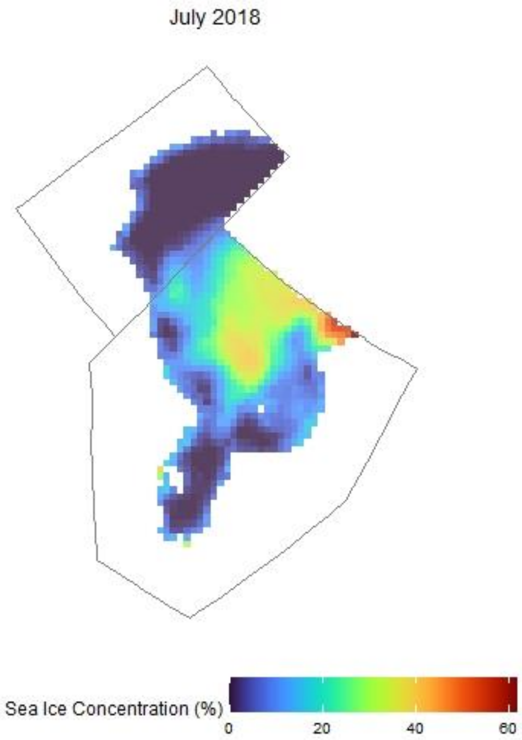
July 2014




July 2015


July 2016


July 2017


July 2018

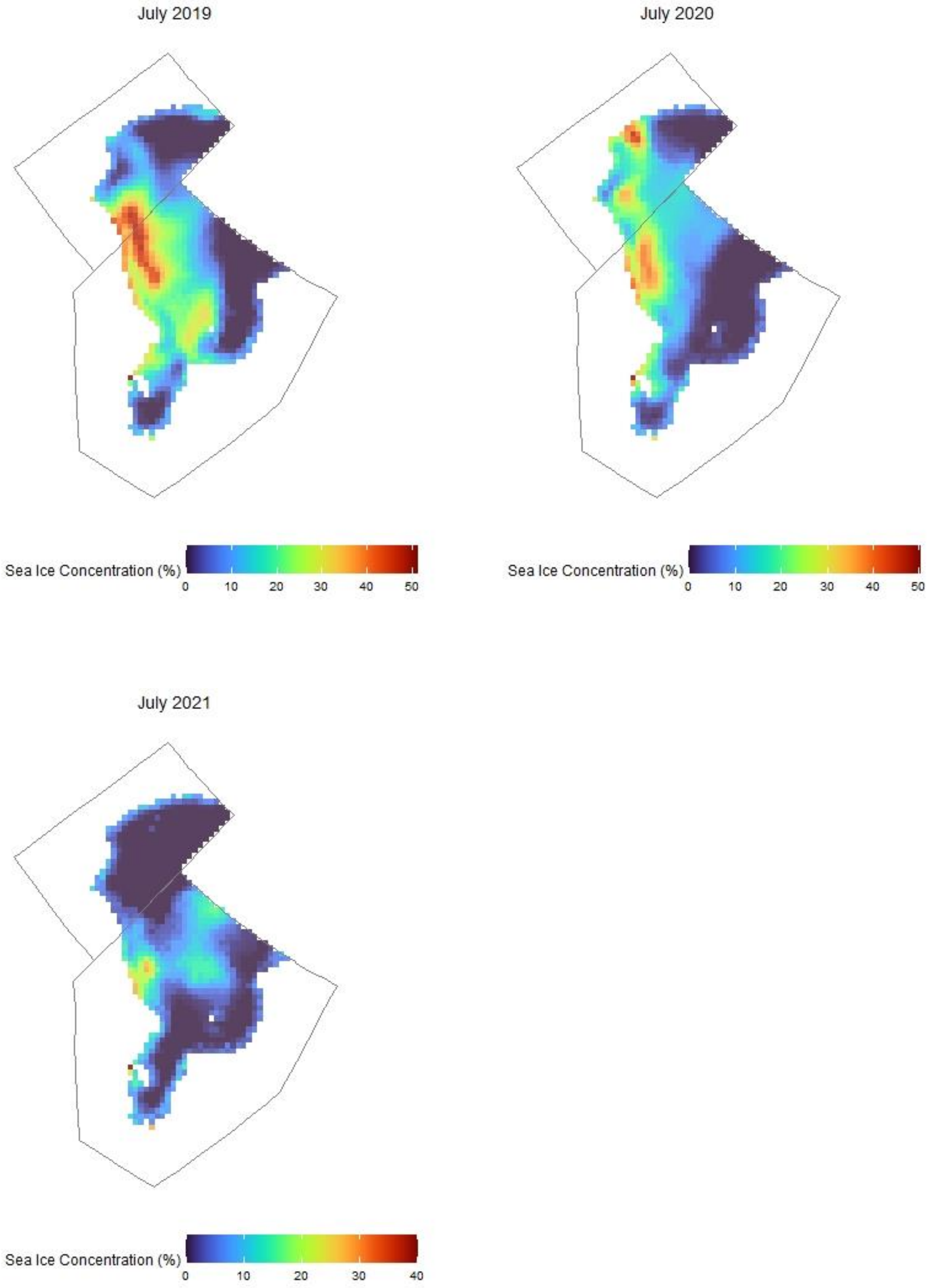


Figure 12. Average sea-ice concentration from July 1 through July 31 for each year from 2011 through 2021 for the Western and Southern Hudson Bay polar bear subpopulations.

These results have complex implications for harvest management. It is our opinion that the increase in SH is due to a combination of reduced harvest mortality during 2016-2021 relative to the 2010-2015 period and improved reproductive output due to both lower harvest levels and improved ice conditions along with annual variation in the distribution of bears between SH and WH. Resolving the degree to which each of these factors is at play is critical for harvest management. Harvest levels are set based, in part, on the number of bears within these subpopulations at the time of surveys. If there are large shifts of the broader distribution, abundances can appear higher or lower than the true number of bears available to be harvested in the respective, current subpopulation boundaries. It remains unclear however, whether such shifts in bears during the ice-free season persists through the ice season or if WH bears shift out of SH and closer to their original marking location in WH once they arrive on land the following year. Ongoing genetic biopsy work along the coastal areas of Manitoba and Ontario along with genetic identification of harvested individuals in WH and SH may help provide insight into the seasonal distribution and movements of bears under dynamic sea-ice changes.

Despite the apparent increase in bears in SH from 2016 to 2021, overall, the combined estimate of WH and SH has declined from 2011 through 2016 and appeared to remain stable between 2016 and 2021. Bears in WH and SH have experienced declines in survival and body condition at least partially related to changes in sea ice (Lunn et al. 1997, Obbard et al. 2007, Regehr et al. 2007, Lunn et al. 2016, Obbard et al. 2016, Sciullo et al. 2016) over the last several decades. Further, both subpopulations are experiencing longer ice-free periods than in the 1980s (Stern and Laidre 2016) providing less access for bears to hunt their preferred prey. This research, in conjunction with harvest data showing high relative harvest rates between 2010 and 2015 plus the results of the 2016 surveys showing declines in abundance and low numbers of yearlings in both subpopulations (Dyck et al. 2017, Obbard et al. 2018) appeared to suggest that a decline in abundance was perhaps underway. However, between 2016 and 2021, ice conditions were more favorable for bears, on average, than between 2011 and 2016, with bears often able to remain on the ice into August (Figs. 10, 11 & 12, OMNRF and ECCC unpublished data). These years of relatively good ice conditions, combined with reduced harvest, may have buffered the population against decline. Indeed, in this current survey, reproduction appeared healthy with a high proportion of yearlings and cubs. However, 2021 was one of the shortest ice seasons of the past decade and survival of yearlings and cubs could be impacted. Our post-stratification estimates indicated that 35% of the SH subpopulation consisted of yearlings and cubs of the year. If the short ice season in 2021 equates to low survival of these bears, the current estimate could immediately become overly optimistic. Continued monitoring of reproduction, survival and inter-annual movements within and between both WH and SH will be critical to continue to inform management during the intervals between aerial surveys.

Limitations and caveats

This survey and analyses were designed and completed to allow for direct comparison to the 2016 aerial survey while taking advantage of recent conceptual advances in mark-recapture distance sampling of polar bears to avoid the underestimation of abundance that results from incorrectly assuming perfect detection of bears on or very close to the transect line. These dual estimates could cause confusion, so we provide rationale for the modelling differences and suggest the most appropriate uses for the different estimates here. In all three years of the SH survey (2011, 2016 and 2021), there were challenges in fitting MRDS models. Specifically, models with distance as a covariate of the mark-recapture submodel counterintuitively did not fit the data well and were not supported by AIC in any of the 3 surveys. Our analysis of data from 2021 suggests that the rear observers' reduced probability of detecting bears near the transect line, such that the overall probability of detecting bears apparently increased with distance near the transect, at least partially explains this lack of fit. Obbard et al. (2018) and our MCDS analyses assumed perfect detection on the transect line. However, these MCDS estimates are negatively biased if bears on the transect line went undetected during the surveys. Modelling imperfect detection on the line (MRDS analyses) yields more accurate estimates if detection probability on the line was < 1.0, and so the best available estimate of SH polar bear abundance in 2021 is the MRDS estimate of 1119 (95% CI 860-1454) bears. Future research should analyze data from all three surveys together using a consistent analytical approach to more formally assess change in bear numbers over time.

In addition to the above caveat, the three SH surveys show that there is likely some underestimation in our distance sampling estimate. In each of the three surveys, the estimate of abundance that combines the distance sampling estimate excluding the coastal zone with the double-observer mark-recapture estimate for the Ontario mainland, coastline and Akimiski island area (lines 2 and 5 in table 5 for the current study) produced a larger abundance estimate than that of the distance sampling estimate alone (lines 1 and 4 in table 5 for the current study). In theory, these estimates should be identical because the total area included in each estimate is the same, only the method used to sample and estimate bear numbers within the coastal zone are different. However, in the 2011 survey, the estimate combining the distance sampling and coastal mark-recapture surveys was 189 bears higher (20% of the final averaged estimate), in 2016 it was 33 bears higher (4% of the final averaged estimate) and in 2021 it was 171 bears higher in the MCDS estimate (17% of the estimate) and 274 bears higher in the MRDS estimate (24% of the estimate) compared to the estimate based solely on distance sampling. We attribute these differences to the highly clustered nature of bear distribution along the coast, which lends itself to high sampling variability. This proposition is supported by our sex and age class results; we estimated through post stratification that there were 173 adult male bears in the Ontario mainland, coastline and Akimiski Island area (95% CI 79-378) when using the distance sampling survey including the coastal zone but saw 184 adult male bears during the coastal mark-recapture survey. These numbers indicate that our point-estimate of adult male bears from the distance sampling portion of the survey was an underestimate, and because adult males concentrate along the coast in large aggregations, we believe the spatial heterogeneity of this class of bears along the coast is the driving cause. This logic would also suggest that our averaged estimates (lines 3 and 6 in table 5) are likely underestimates of the total number of bears in the subpopulation and was likewise an underestimate in 2011 and a smaller underestimate in 2016. The differences across years also matches well with the evidence that bears are displaying substantial variation in their distribution from year to year. Male bears are likely the least philopatric to their summering areas because they do not need to access known inland areas for denning. Thus, if as theorized, the ice conditions in 2011 and 2021 were conducive to greater numbers of bears in SH, with fewer bears in 2016, we assume that most of these bears would be adult males, concentrating along the coast and leading to the larger differences in the estimates in 2011 and 2021 relative to 2016.

Abundance estimate of SH

In light of the above discussion of limitations, the best available evidence indicates that using the most up-to-date modeling approach, the best estimate of the number of bears present in SH in fall 2021 was 1119 (95% CI 860-1454) bears.

Conclusion

Management of polar bears in Canada makes an implicit assumption that subpopulations are discrete units. Surveys are conducted within the boundaries of subpopulations, and quotas are subsequently developed based on those results, with bears only counted against a quota if they are harvested within the bounds of a subpopulation. Although this assumption is almost certainly violated to some degree in every subpopulation, the implications for sustainable harvest of polar bears likely varies greatly depending on the degree of interchange between subpopulations that occurs when surveys to update estimates of abundance are undertaken. As first proposed by Prevett and Kolenosky (1982), our results, combined with those of Atkinson et al. (2022) and ECCC unpublished data suggest that, at least in some years, there is the potential for significant distributional shifts across the boundary between WH and SH. Therefore, these subpopulations are not acting as discrete units, which raises significant challenges for developing quotas based on management boundaries. Further complicating this issue is that much of the WH harvest occurs during the ice-free season when bears are onshore, whereas the majority of SH harvest is on the sea ice (Government of Nunavut, unpublished data) when bears from Foxe Basin, SH, and WH are free to mix (Peacock et al. 2010). In addition, there may be strong demographic differences between these subpopulations. We suggest further research aimed at assessing interannual shifts in distribution, particularly with ongoing climate warming, examining the proportion of bears harvested in subpopulations different from the one they are present in during the survey period and continued monitoring of vital rates in both subpopulations will be key for future management decisions in WH and SH.

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Supplemental material

Table S1. Multiple-covariate distance sampling (MCDS) models, degrees of freedom, Akaike's information criterion (AIC) values and change in AIC from the top model (ΔAIC) for models fit to polar bear distance sampling data collected across the entirety of the Ontario mainland, coastline and Akimiski island area in 2021. Abundance was estimated by model averaging across models marked with asterisks. See main text for description of model structure.

MCDS model	df	AIC	ΔAIC
Half-normal	1	1831.83	0.00
Half-normal + vegetation density*	2	1832.45	0.61
Hazard rate + vegetation density*	3	1832.60	0.76
Half-normal + vegetation height	2	1833.51	1.67
Hazard rate	2	1833.54	1.70
Half-normal + visibility	2	1833.64	1.80
Hazard rate + vegetation height	3	1833.78	1.95
Hazard rate + visibility	3	1835.34	3.50

Table S2. Multiple-covariate distance sampling (MCDS) models, degrees of freedom, Akaike's information criterion (AIC) values and change in AIC from the top model (ΔAIC) for models fit to polar bear distance sampling data collected across the Ontario mainland, coastline and Akimiski island area excluding the coastal zone in 2021. Abundance was estimated by model averaging across models marked with asterisks. See main text for description of model structure.

MCDS model	df	AIC	ΔAIC
Half-normal + visibility*	2	1161.75	0.00
Hazard rate + visibility*	3	1162.84	1.09
Half-normal	1	1164.05	2.30
Hazard rate	2	1164.48	2.72
Hazard rate + vegetation density	3	1165.00	3.25
Half-normal + vegetation density	2	1165.51	3.75
Hazard rate + vegetation height	3	1165.89	4.14
Half-normal + vegetation height	2	1166.05	4.30

Table S3. Mark-recapture distance sampling (MRDS) models, degrees of freedom (df), Akaike's information criterion (AIC) values, difference in AIC from the top model (ΔAIC) and model weights (w_i) used in model averaging for models fit to polar bear distance sampling data collected across the entirety of the Ontario mainland, coastline and Akimiski island area in 2021. All models were included when model-averaging to estimate abundance. We use the top model to estimate the number of bears of different ages and sex by post-stratification. See main text for description of model structure.

Mark-recapture submodel	Distance sampling submodel					
Covariates	Key function	Covariates	df	AIC	ΔAIC	w_i
Blind spot + observer × side + visibility + glare	Half-normal	Vegetation + glare	11	2113.29	0.00	0.30
Blind spot + observer × side + visibility + glare	Hazard rate	Vegetation + glare	12	2114.60	1.32	0.16
Blind spot + observer + side + visibility + glare	Half-normal	Vegetation + glare	10	2114.68	1.39	0.15
Blind spot + observer × side + visibility + glare	Half-normal	Glare	10	2114.87	1.59	0.14
Blind spot + observer + side + visibility + glare	Hazard rate	Vegetation + glare	11	2115.99	2.71	0.08
Blind spot + observer + side + visibility + glare	Half-normal	Glare	9	2116.26	2.98	0.07
Blind spot + observer × side + glare	Half-normal	Vegetation + glare	9	2116.55	3.26	0.06
Blind spot + observer × side + glare	Hazard rate	Vegetation + glare	10	2117.86	4.57	0.03
Blind spot + observer × side + glare	Half-normal	Glare	8	2118.13	4.84	0.03

Table S4. Mark-recapture distance sampling (MRDS) models, degrees of freedom (df), Akaike's information criterion (AIC) values, difference in AIC from the top model (ΔAIC) and model weights (w_i) used in model averaging for models fit to polar bear distance sampling data collected across the Ontario mainland, coastline and Akimiski island area excluding the coastal zone in 2021. All models were included when model-averaging to estimate abundance. See main text for description of model structure.

Mark-recapture submodel	Distance sampling submodel					
Covariates	Key function	Covariates	df	AIC	ΔAIC	(w_i)
Blind spot + observer × side	Half-normal	Visibility	8	1359.18	0.00	0.51
Blind spot + observer × side	Hazard rate	Visibility	9	1360.53	1.36	0.26
Blind spot + observer × side	Half-normal	None	7	1362.06	2.88	0.12
Blind spot + observer × side	Hazard rate	None	8	1362.41	3.23	0.10

Table S5. Estimates of polar bear abundance within the Ontario and Akimiski island coastal zone, obtained using double-observer mark-recapture methods along a coastal transect, proportion of cubs, yearlings and adults for the 4 lasts of surveys conducted in this portion of SH.

Year	Abundance estimate (95% CI) coastal transect	Proportion cubs observed coastal transect	Proportion yearlings observed coastal transect	Proportion adults observed coastal transect
2011	422 (381 – 467)	0.15	0.12	0.60
2016	269 (244 – 297)	0.17	0.03	0.71
2018	249 (230 – 270)	0.10	0.07	0.74
2021	335 (309 – 363)	0.10	0.12	0.67

Consultation with Sanikiluaq Hunters and Trappers Association on the 2021 Southern Hudson Bay Polar Bear Subpopulation Aerial Abundance Survey Report

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ᐊᐅᓚᑦᑎᔨᓂᙶᖅᑐᖅ ᓇᐃᓈᖅᓯᒪᔪᑦ

ᓄᓇᕗᑦ ᒐᕙᒪᖓ, ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑭᒡᒐᖅᑐᖅᑎᖏᑦ ᑲᑎᒪᖃᑎᖃᓚᐅᖅᑐᑦ ᓴᓂᑭᓗᐊᑉ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᖏᓐᓂᑦ ᒪᐃ 13, 2024−ᒥᑦ.

ᑲᑎᒪᑎᑦᑎᔪᒪᓚᐅᖅᐳᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᒻᒪ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᖃᐅᓯᖃᕈᓐᓇᕐᑯᓪᓗᒋᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓂᒃ 2021 ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᐃᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ, ᐊᐅᓚᑕᐅᓪᓗᓂ ᐋᓐᑎᐅᕆᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓴᑦ ᐊᒻᒪ ᓇᐹᖅᑐᓕᕆᓂᖅ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᖃᑎᒋᔭᐅᓪᓗᑎᒃ. 2021−ᒥ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᖅᓱᖅᑕᐅᓚᐅᖅᑐᑦ. ᓯᕗᓪᓖᑦ ᐊᒥᓲᓂᖏᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᐃᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᑲᑎᖅᓱᖅᑕᐅᓚᐅᖅᑐᑦ 2011−ᒥᑦ ᐊᒻᒪ 2016−ᒥᑦ. 2021−ᒥ ᖃᐅᔨᓴᕈᑕᐅᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᓚᐅᖅᐳᑦ ᐅᓄᖅᓯᒋᐊᕐᓂᖏᓐᓂᒃ ᓇᓄᐃᑦ 2016−ᒥ 2021−ᒧᑦ, ᑭᓯᐊᓂ ᖃᐅᔨᓴᕈᑎᕕᓃᑦ ᑲᑎᑕᐅᑎᓪᓗᒋᑦ 2021−ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐊᑭᓐᓇᖓᓂ ᓇᓄᐃᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓂᒃ, ᐊᔾᔨᒌᙱᓐᓂᖃᓚᐅᖅᐳᑦ 1 ᓇᓄᐃᑦ 2016 ᐊᒻᒪ 2021 ᐊᑯᓐᓂᖓᓂ. ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕈᓰᑦ ᑭᒡᓕᖃᕐᓂᖏᓐᓄᑦ, ᖃᐅᔨᓴᐅᔾᔪᓐᓇᓚᐅᙱᓚᑦ ᖃᓄᑎᒥ ᐊᑭᑦᑐᓇᐊᓚᐅᕐᒪᖔᑦ ᐃᖏᕐᕋᓂᖏᓐᓄᑦ. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ (ECCC) ᖃᐅᔨᓴᖃᑦᑕᖅᓯᒪᕗᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᑕᐃᒪᙵᑦ 1980−ᐊᑐᓚᐅᖅᑎᓪᓗᒋᑦ, ᐃᓚᒋᔭᐅᓪᓗᓂ ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᓂᕐᒧᑦ ᓇᐅᒃᑲᖅᑕᐅᑎᓄᑦ ᐱᓕᕆᐊᖅ 2017−ᒥ ᒫᓇᒧᑦ ᑎᑭᖦᖢᒍ. ᓴᖅᑭᑦᑐᑦ ᑖᒃᑯᓇᙵᑦ ᐱᓕᕆᐊᓂᒃ ᐅᖃᓚᐅᖅᐳᑦ ᐅᓄᖅᓯᕚᓪᓕᖅᓯᒪᓂᖏᓐᓂᒃ ᐱᔾᔪᑕᐅᓪᓗᑎᒃ ᐃᖏᕐᕋᓂᖏᑦ ᑭᒡᓕᖓᑕ ᖃᓂᒋᔮᓂ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂ ᐊᒻᒪ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 2021−ᒥ, ᐃᒪᓐᓇᐅᙱᖔᖅᑐᖅ ᐃᕐᓂᐅᖅᐸᒃᑐᑦ ᐅᓄᕐᓂᖅᓴᐅᓂᖏᑦ ᒪᓕᒃᑕᐅᙱᖔᖅᖢᑎᒃ. ᑐᕌᕆᔭᐅᔪᑦ ᐅᖃᖃᑎᒌᖕᓂᕆᔭᐅᔪᓂ ᐅᓂᒃᑲᐅᓯᕆᔭᐅᔪᓂᒡᓗ ᐱᑎᑦᑎᔪᒪᓂᐅᓚᐅᖅᐳᑦ ᐊᐅᓚᑦᑎᖃᑎᖃᕐᓂᕐᒧᑦ ᐱᓕᕆᖃᑎᒋᔭᐅᕙᒃᑐᓂᒃ ᖃᓄᐃᓕᖓᓂᖏᓐᓂᒃ ᖃᓄᐃᓕᖓᓕᕐᓂᕆᔭᐅᕗᑦ ᐅᓂᒃᑲᐅᓯᕐᓂ ᐊᒻᒪ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᖓᑦᑎ, ᑲᑎᖅᓱᖅᖢᒋᑦ ᑭᐅᔾᔪᑎᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᐅᖃᑦᑕᖅᑐᓂᒃ, ᑲᑎᖅᓱᖅᖢᒋᓪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᒃᑲᓐᓂᖏᑦ, ᐅᖃᐅᓯᕆᓗᒍᓗ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᖑᓂᕐᒧᑦ ᐊᑐᓕᖅᑕᐅᕐᑯᓯᒪᔭᖏᑦ.

ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᐲᑦ ᑲᑎᖃᑎᖃᓚᐅᖅᐳᑦ ᓴᓂᑭᓗᐊᕐᒥᐅᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᖏᓐᓂᒃ, ᑖᒃᑯᐊᓗ ᐊᖑᓇᓱᒃᐸᒃᖢᑎᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᐊᑕ ᓇᓄᖏᓐᓂᑦ.

ᐃᑲᔪᖅᓯᐃᓂᖅ ᐊᑭᕋᖅᑐᕐᓂᕐᓘᓐᓃᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᐊᑐᓕᕐᑯᔭᖓᓂᒃ ᐅᖃᐅᓯᐅᓪᓚᕆᓚᐅᙱᓚᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓂᑦ ᑐᓴᕋᓱᒃᑎᓪᓗᒋᑦ. ᑐᓴᕋᓯᐊᓂᒃᑎᓪᓗᒋᑦ, ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐱᖅᑯᑎᒃᓴᖅᑎᑕᐅᓚᐅᖅᐳᑦ ᐅᖃᐅᓯᖃᕈᒪᓪᓗᑎᒃ ᐊᒻᒪ ᐃᓱᒪᒋᔭᒥᓂᒃ ᓇᒃᓯᐅᔾᔨᓪᓗᑎᒃ ᐊᕙᑎᓕᕆᔨᒃᑯᓄᑦ ᐅᕙᓘᓐᓃᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ, ᓇᓕᐊᑐᐃᓐᓇᖅ ᓂᕈᐊᕆᔭᐅᓚᐅᖅᑐᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓄᑦ.

ᑭᐅᔾᔪᑕᐅᔪᑦ ᐊᒻᒪ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᑲᑎᑕᐅᔪᑦ ᑖᒃᑯᓇᓂ ᑐᓴᕋᓱᒃᑎᓪᓗᒋᑦ ᐃᓱᒪᒋᔭᐅᓂᐊᖅᐳᑦ ᐋᖅᑭᒃᓯᓕᖅᐸᑕ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐊᑐᓕᕐᑯᓯᔾᔪᑎᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᕐᓄᑦ ᑐᓐᓂᖅᑯᑕᐅᓛᖅᑐᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᖁᓪᓗᒍ.

ᐅᓇ ᐅᓂᒃᑳᖅ ᓇᐃᓪᓕᒋᐊᕆᓯᒪᔪᑦ ᐅᖃᐅᓯᐅᑕᐅᖅᑐᓂᑦ ᑐᓴᕋᓱᒃᑎᓪᓗᑕ.

ᐱᒋᐊᕐᓂᖓ

ᐅᓇ ᐅᓂᒃᑳᖅ ᐊᑭᑎᓕᕆᔩᒃᑯᑦ ᐱᓇᓱᑦᑎᐊᕈᓐᓇᕐᓂᓕᒫᖏᓐᓂᑦ ᑕᒻᒪᖅᓯᒪᙱᑦᑐᓂᑦ ᐃᓕᐅᖅᑲᐃᓇᓱᒃᓯᒪᔪᑦ ᑐᓴᖅᑕᐅᓯᒪᔪᓕᒫᓂᑦ ᑐᓴᕋᓱᒋᐊᖅᑎᓪᓗᒋᑦ ᑲᑎᒪᓪᓗᑎᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᓐᓂᑦ ᓴᓂᑯᓗᐊᕐᒥᑦ ᒪᐃ 13, 2024−ᒥᑦ.

ᐃᓱᒪᒋᔭᖏᑦ ᓴᖅᑭᑕᑦ ᐅᕙᓂ ᐊᔾᔨᒋᓗᐊᒻᒪᓇᙱᓚᖏᑦ ᐊᑭᑎᓕᕆᔩᒃᑯᑦ, ᓄᓇᕗᒻᒥᒎᓐᓃᑦ ᑲᕙᒪᒃᑯᓐᓂᑦ.

ᐃᓗᓕᖏᑦ

1.0 ᐅᓂᒃᑳᑉ ᐱᔾᔪᑖ ᐋᖅᑭᒃᓯᒪᓂᖓᓗ

ᑖᓐᓇ ᐅᓂᒃᑳᖅ ᑲᑎᖅᓱᐃᔾᔪᑕᐅᓇᓱᐊᖅᐳᖅ ᐅᖃᐅᓯᒃᓴᕆᔭᐅᔪᓂᒃ, ᐊᐱᖅᑯᑕᐅᔪᓂᒃ, ᐃᓱᒫᓗᑕᐅᔪᓂᒃ ᖃᓄᐃᓕᐅᕈᑕᐅᖁᔭᐅᔪᓂᓪᓗ ᓴᓂᑭᓗᐊᕐᒥᐅᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᖓᓐᓂᑦ ᑭᐊᔾᔪᑕᐅᓗᓂ 2021 ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᕐᑯᑎᖏᓐᓂ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ.

ᑭᒡᓚᖅᑐᐃᓈᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᖕᓂᑦ (GN) ᐃᓚᐅᓚᐅᖅᐳᑦ ᐅᖃᖃᑎᖃᕐᓂᕐᒧᑦ ᐃᓚᖃᖅᖢᑎᒃ ᐊᓯᖏᓐᓂᒃ ᐊᐅᓚᑦᑎᖃᑎᖃᕐᓂᕐᒧᑦ ᐱᓕᕆᖃᑎᒌᓂᒃ ᑕᒪᑐᒧᖓ ᐃᓱᒪᒋᔭᖃᖅᓯᒪᔪᓄᓪᓗ ᐱᓕᕆᖃᑕᐅᔪᓄᑦ.

2.0 ᑐᓴᕋᓱᖕᓂᕐᒧᑦ ᐱᔾᔪᑖ

ᑲᑎᒪᑎᑦᑎᔪᒪᓚᐅᖅᐳᑦ ᐅᖃᐅᓯᖃᕈᒪᓪᓗᑎᒃ ᓄᑖᖑᓖᒥᒃ ᖃᐅᔨᓴᖅᑎᓂᒃ ᑐᑭᓯᒋᐊᒐᒃᓴᑦ ᐱᔾᔪᑎᓖᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᕐᓂᒃ ᐅᖃᐅᓯᐅᓯᒪᔪᓂᒃ 2021−ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕈᑎᑦ ᐅᓂᒃᑳᒥ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑎᓂᒃ ᐱᓕᕆᐊᖑᓯᒪᔪᓂᒃ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓄᑦ ᐊᒻᒪ ᓯᓚ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᔪᓕᕆᔨᒃᑯᓐᓄᑦ ᑲᓇᑕᒥ. ᐊᒻᒪᓗᑦᑕᐅᖅ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦ ᓴᖅᑭᑦᑎᓚᐅᕐᒥᔪᑦ ᑲᑎᖅᖢᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖃᓄᐃᓕᐅᕆᐊᖅᑯᔭᓂᑦ ᑐᓴᕋᓱᒃᑎᓪᓗᒋᑦ, ᑭᓯᐊᓂᓪᓕ ᐅᖃᐅᓯᖃᓚᐅᕐᒥᔪᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᓇᓱᒃᑐᑦ ᐋᖅᑭᒃᑕᐅᔪᓐᓇᕐᓂᖏᓐᓂᑦ ᒪᓕᒡᓗᑎᒃ ᓄᓇᓖᑦ ᐱᔭᕆᐊᓕᖏᓐᓂᑦ ᑎᑭᑕᐅᓇᓱᐊᖅᑐᓂᓪᓗ ᑖᒃᑯᓄᖓ ᓇᓄᕐᓄᑦ.

2.1 ᑲᑎᒪᓃᑦ ᐋᖅᑭᒃᓱᖅᓯᒪᓂᖏᑦ

ᑲᑎᒪᓚᐅᖅᑐᑦ ᐅᓐᓄᒃᑯᑦ ᐃᑲᕐᕌᖕᓄᑦ ᒪᕐᕉᖕᓄᑦ ᓇᑉᐸᕐᓗ. ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᓇᓄᓕᕆᔨ, ᐊᓖᓴ ᐴᕼᐋᑦ, ᐊᒻᒪ ᐆᒪᔪᕐᓂᐊᖅᑎᒃᑯᑦ ᐅᖃᐅᔾᔨᒋᐊᖅᑎᒻᒪᕆᖓᑦ, ᔫᓇᑖᓐ ᐱᓐ, ᐱᓕᕆᑎᑦᑎᓚᐅᖅᐹᒃ ᑲᑎᒪᑎᑦᑎᓪᓗᑎᓪᓗ. ᔩᓯᐱ ᓄᐊᑦᕋᑉ, ᐋᓐᑎᐅᓇᐆᒥ ᓄᓇᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᐊᒻᒪ ᓇᐹᖅᑐᓂᒃ ᐊᕕᒃᑐᖅᓯᒪᔪᓂ ᐆᒪᔪᓕᕆᔨ ᐅᐸᒍᒪᓚᐅᖅᑐᖅ, ᑭᓯᐊᓂ ᐅᐸᒍᑎᓚᐅᙱᑦᑐᖅ ᐃᖅᑲᓇᐃᔭᑎᐊᕐᑐᓂ ᑐᐊᕕᕐᓇᖅᑐᒧᑦ. ᐊᑐᓂ ᐅᓂᒃᑳᖅᑐᑦ ᐱᒋᐊᓚᐅᖅᑐᑦ ᐅᒃᑯᐃᖅᓯᓂᕐᒧᑦ ᐅᖃᐅᓯᒃᓴᓂᑦ ᐊᓖᓴ ᐴᕼᐋᑦ, ᐊᒻᒪ ᔫᓇᑖᓐ ᐱᓐ ᑭᒃᑯᖓᖔᖏᑦ ᑭᓱᓂᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᓂᑦ ᑭᒡᓚᖅᑐᐃᖓᖔᖏᑦ ᐊᒻᒪᓗᑦᑕᐅᖅ ᑐᓴᕋᓱᖕᓂᐅᑉ ᐱᔾᔪᑎᐊᓂᑦ. ᑭᖑᓂᐊᒍᑦ ᖃᓇᑕᐅᔭᒃᑯᑦ ᐅᓂᒃᑳᓕᐊᖑᓚᐅᖅᑐᑦ ᕿᒥᕐᕈᓗᑎᒃ ᖃᐅᔨᓴᖅᑎᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ ᐊᒻᒪ ᐊᐅᓚᑦᑎᖃᑎᒌᖕᓂᕐᒧᑦ ᐱᓕᕆᔾᔪᓯᕐᓂᒃ, ᑕᐃᓱᒪᓂᑕᓴᐃᑦ ᐊᐅᓚᑦᑎᔾᔪᓯᐅᖃᑦᑕᖅᓯᒪᔪᑦ, ᐅᓂᒃᑲᐅᓯᐅᑦᑎᐊᖅᓯᒪᔪᑦ ᖃᓄᐃᓕᖓᓕᖅᓂᓇᓚᐅᖅᑕᖏᑦ 2021−ᒥ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ, ᖃᓄᐃᓕᖓᓕᖅᓂᖏᓪᓗ ᖃᐅᔨᓴᖅᑎᓂᒃ ᐱᓕᕆᐊᖑᓚᐅᖅᑐᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᑕᓯᐅᔭᕐᔪᐊᒦᑦᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᓗᒋᑦ (ᐅᐃᒍᖓ 1). ᖃᐅᔨᓴᕈᑎᕕᓃᑦ ᖃᖓᑕᓲᒃᑯᑦ ᓇᐃᓴᐃᓂᖅᒍᖓᓕᖅᑦ ᐊᔾᔨᒌᓚᐅᙱᑦᑐᑦ, ᖃᐅᔨᓴᖅᑎᒻᒪᕇᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᐃᑲᔪᖅᑕᐅᓪᓗᑎᒃ ᓇᓗᓇᐃᓴᐃᑎᓪᓗᒋᑦ ᖃᖓᑕᓲᒃᑯᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᕕᓂᕐᓂᑦ. ᐃᓚᐅᖃᑕᐅᔪᑦ ᐊᐱᖅᑯᑎᖃᕋᓱᖕᒪᑕ ᐊᐱᕆᖃᑦᑕᖁᔭᐅᓚᐅᖅᐳᑦ, ᐃᓱᒫᓘᑎᒋᔭᒥᖕᓂᑦ ᓂᓪᓕᖃᑦᑕᕐᓗᑎᒃ, ᐅᖃᐅᔾᔭᐃᒋᐊᖅᐸᓪᓗᑎᓪᓗᓈᑦ ᑲᑎᒪᑎᓪᓗᒋᑦ. ᑲᑎᒫᓂᒃᑎᓪᓗᒋᑦ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᖃᓄᐃᓕᖓᓂᖓ ᑲᑎᖅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᐊᓄᑦ ᐅᓂᒃᑲᐅᓯᐅᑕᐅᓚᐅᖅᐳᑦ.

ᐅᓂᒃᑳᖒᓂᒃᑎᓪᓗᒋᑦ, ᐊᐱᖅᑯᑎᑦ/ᐅᖃᐅᓯᒃᓴᑦ ᑲᔪᓯᓚᐅᖅᐳᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᓂᕐᒥ ᓄᖅᑲᕋᓱᖕᓂᖕᒋᓐᓂ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐱᖅᑯᑎᒃᓴᖃᓚᐅᖅᐳᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ ᐊᒻᒪ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᐅᓚᑦᑎᔾᔪᑎᖕᒋᑕ ᑐᕌᒐᖕᒋᓐᓂᒃ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂ ᐱᓕᕆᕝᕕᖕᒥᙶᖅᑐᓂᒃ ᐅᖃᐅᓯᖃᖅᓱᑎᒃ ᑭᐅᓪᓗᑎᒡᓗ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖕᒋᓄᑦ.

3.0 ᓇᐃᓈᖅᓯᒪᔪᑦ ᑐᓴᕋᓱᒍᑕᐅᓚᐅᖅᑐᑦ

ᑖᓇ ᐅᓂᒃᑳᖅ ᑲᑎᙵᑎᑦᑎᕗᖅ ᐊᑐᖅᑕᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᐅᔪᓂᑦ. ᐅᖃᐅᓯᐅᓯᒪᓕᕐᓄᑦ ᐃᓚᐅᖃᑕᐅᓯᒪᓕᕐᓄᓪᓗ ᑲᑎᒪᓂᐅᔪᓂᑦ, ᑕᑯᓗᒍ ᐅᐃᒍᖓ A.

ᐅᖃᐅᓯᒃᓴᑦ ᐊᐱᖅᑯᑎᒃᓴᓪᓗ:

- ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐱᖅᑯᑎᒃᓴᖃᓚᐅᖅᐳᑦ ᐃᓱᒫᓗᑎᖃᖅᓱᑎᓪᓗ ᓇᓄᐃᑦ ᖃᓄᐃᙱᑦᑎᐊᕐᓂᖕᒋᓐᓂᒃ ᑎᒥᖕᒋᓐᓂᑦ ᐲᖅᓯᖃᑦᑕᕐᓂᕐᒧᑦ ᐊᒻᒪ ᕿᖕᒐᓯᕈᖅᓯᓂᕐᒧᑦ/ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᔪᓄᑦ.
- ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᓇᓄᖃᕐᔪᐊᓚᐅᕐᓂᖓᓂᒃ 2021–ᒥᑦ, ᒥᐆᑦ ᐃᕙᕝᕕᖕᒋᓐᓂᒃ ᓱᕋᒃᑎᕆᓪᓗᑎᒃ. ᑕᒪᓐᓇ ᒪᓕᓚᐅᖅᐳᖅ ᖃᐅᔨᓴᐅᔪᓄᑦ ᐊᒥᓲᓂᖅᓴᓄᑦ ᓇᓄᕐᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ 2021–ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᓚᐅᖅᑐᒥ.
- ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑐᓴᖅᑎᑦᑎᓚᐅᖅᐳᑦ ᓯᑯ ᓵᒡᓕᕗᓪᓕᐊᓕᕐᓂᖓᓂᒃ ᐊᒻᒪ ᓇᓚᐅᑦᑖᓗᒃᓴᐅᔪᓐᓃᖅᓱᓂ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓ.
- ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐱᔪᒪᓂᖕᒋᓐᓂᑦ ᓇᒃᓯᐅᔾᔨᓗᑎᒃ ᓇᖕᒥᓂᖅ ᑎᑎᖅᑲᒥᑦ ᓄᓇᕗᒻᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖕᒋᓄᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᖑᓇᓯᒃᓴᓄᑦ ᐃᓱᒪᓕᐅᕆᓂᐊᕐᓗᑎᒃ ᐅᑐᐱᕆᒥᑦ ᑲᑎᒪᓂᕐᒥᑦ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᑲᔪᖅᑐᐃᓚᐅᙱᑦᑐᑦ ᐃᑲᔪᖅᑐᐃᓚᐅᙱᑦᑐᓪᓘᓐᓃᑦ ᑎᑎᕋᖅᑕᐅᙵᖅᑐᒥᑦ ᑲᑎᕐᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᓇᓱᖅᑐᓂᑦ ᓴᖅᑭᑕᐅᔪᓂᑦ ᓄᓇᕗᑦ ᓚᕙᒃᑯᖕᒋᓄᑦ.

ᐅᐃᒍᖓ A

2021 ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᖕᒋᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐅᓂᒃᑳᖅ ᐅᖃᖃᑎᖃᕐᓂᕐᒧᑦ ᑲᑎᒪᓂᕐᒥ ᐅᖃᐅᓯᐅᓪᓗᐊᑕᖅᓯᒪᔪᑦ

ᓇᓂ: ᕿᑭᖅᑕᐃᑦ ᐃᒡᓗᖓ (ᐅᑭᐅᖅᑕᖅᑐᒥ ᒥᑎᕐᓄᑦ ᑲᑐᔾᔨᖃᑎᒌᑦ/ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᒡᓗᖅᑯᑖ)
ᐅᓪᓗᖅ/ᓇᒧᙵᖅᑎᓪᓗᒍ: ᒪᐃ 13, 2024
ᖃᐅᔨᓴᐅᑦ ᐱᒋᐊᕐᕕᖓ: 19:00
ᐃᓱᓕᕝᕕᖓ: 21:30

ᐃᓚᐅᔪᑦ

ᐃᓚᐃ ᑲᕕᒃ (ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐅᓚᑦᑎᔨᒋᓚᐅᑲᒃᑕᖓ)
ᒥᒃ ᐊᑉᐸᖅ (ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᒃᓯᕙᐅᑕᐅᑉ ᑐᓪᓕᐊ)

ᓵᓕ ᑕᑲᑕᒃ (ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑎᑎᕋᖅᑎᖓ/ᑮᓇᐅᔭᓕᕆᔨ)
ᒨᓯᓯ ᐊᑉᐸᖃᖅ
ᒋᐅᓕ ᕋᒻᐆᑦ
ᔫᓂ ᑯᑦᑐᐊᕈᒃ
ᐊᓖᓴ ᐆHᐋᕐᑦ, ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑎ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᖕᒋᓐᓂᑦ
ᔫᓇᑕᓐ ᐱᓐ, ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᐅᖃᐅᔾᔪᐃᔨᒻᒪᕆᒃ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ
ᑲᐃᐅᓪ ᕆᑦᓯ, ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᐊᑐᐊᒐᓕᕆᓂᕐᒧᓪᓗ ᐅᖃᐅᔾᔨᓯᒋᐊᖅᑎ, ᐆᒪᔪᓕᕆᔨᒃᑯᓐᓂ ᐆᒪᔪᓕᕆᔨ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ
ᔩᓇ ᕿᑦᑐᓱᒃ, ᐆᒪᔪᕐᓂᐊᖅᑎ, ᓴᓂᑭᓗᐊᖅ
ᓕᓯ ᖃᕝᕕᒃ-ᒥᑭᔪᖅ, ᑐᓵᔨ
ᐅᐸᒍᑎᓚᐅᙱᑦᑐᑦ: ᔩᓯᐱ ᓄᐊᑦᕋᑉ, ᐊᕕᒃᑐᖅᓯᒪᔪᓂᒃ ᐆᒪᔪᓕᕆᔨ, ᐋᓐᑎᐅᕆᐅᒥ ᓄᓇᒥᐅᑕᐃᑦ ᐊᑐᖅᑐᒃᓴᑦ ᐊᒻᒪ ᓇᐹᖅᑐᓕᕆᔨ

ᑐᕌᒐᖅ ᑲᑎᒪᓂᕐᒧᑦ:

ᐊᑐᐃᓐᓇᕈᖅᑎᓪᓗᒋᑦ ᐅᓂᒃᑲᐅᓯᐅᔪᑦ ᓄᑖᖑᓛᑦ ᖃᐅᔨᓴᖅᑎᑦ ᑐᑭᓯᒋᐊᒐᒃᓴᖕᒋᑦ ᐱᔾᔪᑎᓖᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᓇᓄᕐᓂᒃ ᐅᓂᒃᑲᐅᓯᐅᓯᒪᔪᓂᒃ 2021-ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᑯᖕᒋᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᐅᓂᒃᑳᒥ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑎᓄᑦ ᐱᓕᕆᐊᖑᓂᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ. ᐅᖃᖃᑎᒌᒍᑎᒋᓗᒋᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᑦ ᑲᑎᔅᓱᒋᑦ ᖃᔅᓯᕌᕈᓐᓇᖅᑎᑕᐅᓂᕐᒧᑦ ᐱᕐᑯᔭᐅᔪᖅ. ᐅᖃᖃᑎᒌᒍᑎᒋᓗᒋᑦ ᖃᓄᐃᓐᓇᖅ ᐊᐱᖅᑯᑎᒃᓴᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑕᒪᐃᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ. ᑲᑎᖅᓱᕐᓗᒋᑦ ᐅᖃᐅᓯᒃᓴᓇᔭᐅᔪᑦ ᐊᒻᒪ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᐊᐅᓚᑦᑎᔾᔪᑎᖕᒋᓐᓄᑦ.

ᐊᑐᖅᑕᐅᓂᐊᓕᖅᑐᑦ ᖃᓄᐃᓕᐅᕈᑕᐅᓂᐊᖅᑐᓪᓗ:

ᕿᓚᒥᐅᔪᒃᑯᑦ ᖃᓄᐃᓕᐅᕈᑏᑦ (ᐱᓇᓱᐊᕈᓰᑦ 4–6 ᐃᓗᐊᓂᑦ):

- ᐃᖕᒌᓐᓈᖅᑕᐅᒃᑯᑦ ᑎᑎᕋᖅᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐱᖅᑯᑎᖕᒋᑦ

ᐊᑯᓂᐅᔪᕐᒧᑦ ᖃᓄᐃᓕᐅᕈᑏᑦ (ᑕᖅᑮᑦ ᒪᕐᕉᒃ ᐱᖓᓱᑦ ᐊᑐᕐᓂᐊᖅᑐᓂᒃ):

- ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᓇᓚᐅᑦᑖᖅᑎᓪᓗᒋᑦ ᖁᖓᓯᕿᖅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ

ᑲᑎᒪᔾᔪᑏᑦ ᐱᔾᔪᑕᐅᓗᐊᖅᓯᒪᔪᑦ:

ᐊᐱᖅᑯᑎ ᑲᑎᒪᔪᓄᑦ ᓇᓗᓇᐃᔭᖅᑕᐅᓚᐅᖅᑎᓪᓗᒍ:

ᑲᑎᒪᓂᖅ ᐱᒋᐊᓚᐅᖅᑐᖅ ᓇᓗᓇᐃᖅᓯᓪᓗᑎᒃ ᐃᓚᐅᔪᓂᒃ. **ᔩ ᓄᐊᑐᕋᑉ** ᐸᕐᓇᐃᓯᒪᓚᐅᕋᓗᐊᖅᓱᓂ ᐅᐸᖕᓂᐊᕆᐊᒃᓴᖅ, ᑭᓯᐊᓂ ᑐᐊᕕᕐᓇᖅᑐᒧᑦ ᐃᓚᐅᔪᓐᓇᓚᐅᙱᒻᒪᑦ. **ᐊᓖᓴ ᐆᕐHᐋᑦ** ᑐᓴᖅᑎᔅᓱᓂᒋᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐱᖅᑯᑎᒃᓴᖅᐸᑕ ᑭᐅᔭᓐᓇᙱᑕᒥᓂᒃ, ᔫᒧᑦ ᑐᓂᓇᔭᖅᑕᖕᒋᑦ. **ᐊᓖᓴ ᐆᕐᑦ** ᐅᓂᒃᑲᐅᓯᖅᑲᓚᐅᖅᐳᖅ ᐊᔾᔨᙳᐊᒥᑦ ᖃᓄᐃᓕᐅᖅᑲᑦᑕᖅᓯᒪᓂᖕᒋᓐᓂᑦ ᓃᐸᓪᓕᐊᓂᖕᒋᓐᓂᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖕᒋᑦᑕ ᓇᓄᕐᓂᑦ ᖃᐅᔨᓴᖅᑎᖕᒋᓐᓂᑦ ᐆᒪᔪᓕᕆᔨᒥᑦ ᔭᒐᐃᓚᐅᕐᓂᖕᒐᓄᑦ ᒫᑲᔅ ᑎᒃᒥᑦ. ᐅᓂᒃᑲᐅᓯᖅᑲᓕᓚᐅᕐᒥᔪᖅ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖕᒋᑕ ᒥᒃᓵᓄᑦ ᐊᒻᒪᓗ ᐸᕐᓇᖕᓂᐅᑉ ᓇᓃᓕᕐᒪᖕᒑᑦ ᑎᑭᓪᓗᒍ ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᐅᓚᑕᐅᓂᖕᒋᓐᓄᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᔪᖅ ᐊᖏᖅᑕᐅᒃᐸᑦ ᒥᓂᔅᑕᒧᑦ ᐊᒻᒪ ᓇᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᓂᕕᙵᐅᒃᑯᑕᖏᑦ ᑐᓂᐅᖅᑲᑕᐅᕙᒃᓱᑎᒃ ᓄᓇᓕᖕᓄᑦ. ᐅᖃᓕᕆᓪᓗᓂ ᐱᔾᔪᑎᖓ ᑖᔅᓱᒪ ᑲᑎᒪᓂᐅᑉ ᐊᑐᐃᓐᓇᕈᖅᑎᑦᑎᔪᒪᖅᑲᐅᓂᖕᒋᓐᓂᒃ ᓇᐃᓈᖅᓯᒪᔪᓂᒃ ᖃᓄᐃᓕᖓᓕᖅᑲᑦᑕᖅᓯᒪᔪᓂᒃ ᖃᖓᑕᓲᒃᑯᑦ ᖃᔅᓯᐅᓂᖅᑲᖓᓂ ᖃᐅᔨᓴᖅᑎᖕᒋᓐᓄᑦ ᐊᒻᒪ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᐱᓕᕆᐊᖕᒋᓂᒃ ᑐᓴᕐᓂᐊᕈᒪᓗᒋᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑭᐅᔾᔪᑎᒃᓴᖏᑦ.

ᖃᓪᓚᑦᓱᒃᑯᑦ ᖃᓄᖅ ᐅᓄᖅᑎᒋᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕈᑎᕕᓂᑦ ᓴᖅᑭᑎᑕᒫᐃᑦ

- ᐅᖃᐅᓯᒃᓴᖃᖅᑐᖃᓛᐅᙱᑦᑐᖅ.

ᖃᓄᖅ ᐅᓄᖅᑎᒋᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕐᓂᖅ/ᓯᓚᒥ ᐱᓕᕆᓂᖅ

- **ᒨᓯᓯ:** ᑐᑭᓯᔪᓐᖓ ᐊᒥᓲᓂᖏᑦ ᓇᓄᐃᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑎᓪᓗᒋᑦ. ᓇᓗᓇᐃᖅᑕᐅᕙᒃᐸᑦ ᐊᖑᑕᐅᒃᐸᑕ ᐊᕐᓇᐅᑉᐸᑕᓘᓐᓃᑦ?
- **ᐋᓪᓚ**: ᑕᑯᒃᓴᐅᔪᒥᒃ ᓇᓗᓇᐃᒃᑯᑕᒥᒃ ᐱᖃᑦᑕᖅᑐᑦ. ᑕᑎᖃᖅᐸᒃᑭᕗᒍᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔾᑯᓂᑦ ᐅᖃᐅᔾᔭᐅᔭᒪᓪᓗᑕ ᓇᓄᐃᑦ ᑕᑯᔭᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐊᖑᑕᐅᖕᒪᖕᒫᑕ ᐊᕐᓇᐅᖕᒪᖕᒫᑕᓘᓐᓃᑦ, ᑭᓯᐊᓂ ᓇᓗᓇᐃᕈᓐᓇᙳᕆᓚᕗᑦ ᑎᒥᖏᑎᒍᑦ. ᓇᓚᐅᑦᑖᖅᓯᒪᔫᔭᖅᑐᖅ.
- **ᓅᓕ:** ᐃᑲᔪᓚᐅᖅᑐᖓ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ ᐊᒻᒪ ᓇᓗᓇᕐᓂᖅᓴᐅᓲᖑᖕᒪᑕ ᓇᓗᓇᐃᕆᐊᒃᓴᖅ ᐊᖑᓴᓪᓗᐃᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᕐᓇᓪᓗᐃᑦ ᖃᓪᓚᑎᓪᓗᒋᑦ ᓄᓇᒦᕐᔪᓂᓕ ᓇᓗᓇᖅᐸᙱᒻᒪᑦ.

- **ᔫᓂ:** ᑕᐃᒃᑯᓄᖓ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓄᑦ, ᖃᐅᔨᒪᔭᐅᕙ ᐅᖓᓱᒃᑐᒧᑦ ᐃᖏᕐᕋᒃᑦᑕᕐᒪᖕᒫᑕ? ᐅᕝᕙᓘᓐᓃᑦ ᐊᔾᔨᐸᓗᖏᓐᓃᓪᓗᑎᒃ ᖁᖓᓯᕈᖅᓯᒪᖃᑕᐅᔪᓂᑦ?
- **ᐋᓪᓚ**: ᑐᒥᖏᑎᒍᑦ ᑕᑯᓯᒪᔭᕗᑦ, ᐅᖓᓱᒃᑐᓕᐊᕐᔪᐊᖃᑦᑕᕐᒪᑕ, ᓇᐅᒃᑯᕈᓘᔫᓗᒃ ᑕᓯᐅᔭᕐᔪᐊᒥ. ᐱᓕᕆᔪᒧᑦ ᑖᒃᑯᐊ ᑐᓴᖅᑕᐅᔫᒃᓴᑦ ᑎᑎᖅᑐᕆᕈᓇᓲᕐᓗᑎᒃ ᐊᔪᙱᑦᑐᓄᑦ ᖁᖓᓯᕈᕐᒥ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᓂᒃ. ᑖᕆᔫᒃᓴᖃᕐᑦᑕ ᓇᓄᕐᓂᒃ ᖁᖓᓯᕈᖅᑕᐅᓚᐅᖅᑐᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖓᓂᑦ ᐊᒻᒪ ᑐᓴᕋᓯᓚᐅᖅᑎᓪᓗᒋᑦ ᑕᑯᑎᑦᑐᓇᖅᑕᕗᑦ ᑖᕆᔫᒃᓴᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔾᑯᑦ ᐱᔪᒪᒃᐸᑕ.
- **ᓅᓕ:** ᐃᓚᐅᖃᑕᐅᓪᓗᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒧᑦ, ᐋ ᓇᓄᐃᑦ ᓄᓇᒥ ᓇᓗᓇᐃᖅᑕᐅᔫᓇᓚᐅᖅᑐᑦ, ᑭᓯᐊᓂ ᑕᑯᔭᒃᓴᐅᑦᑎᐊᓚᐅᙱᑦᑐᑦ ᐃᒫᓂ. ᑕᑯᓚᐅᖅᑐᒍᑦ ᐃᓚᖏᓐᓂᒃ ᓇᓄᕐᓂᒃ, ᑭᓯᐊᓂ ᖃᐅᔨᓚᐅᖅᑰᙳᓐᓇᑦᑕ ᓇᓄᕐᓂᒃ ᐃᒫᓃᑦᑐᓂᒃ ᑕᑯᔭᒃᓴᐅᓚᐅᙱᓐᓂᖏᓐᓄᑦ.
- **ᐋᓪᓚ**: ᐋ, ᑕᒪᓐᓇ ᑐᓴᖅᓯᒪᖕᒥᔭᕗᑦ. ᐊᔪᕐᓇᕈᔪᖕᒪᑦ ᐊᑦᑕᕐᓇᖅᑕᐃᓕᒪᓂᕐᒧᑦ ᐱᔾᔪᑎᖃᐅᖅᓯᒍᑎᒃ ᐃᒪᐅᑉ ᖁᕐᓵᒍᑦ ᖃᓪᓚᑕᕆᐊᒃᓴᖅ, ᑭᓯᐊᓂᓕ ᖃᓪᓚᑦᑕᖅᑎᑕᓚᒃᓴᒥᒃ ᐊᔾᔨᓕᐅᕈᑎᒧᑦ ᖃᐅᔨᓴᐃᓐᓇᖅᑐᑦ ᑕᒪᑐᒪᓂᓗ ᐊᕐᕌᒍᒥ ᖃᐅᔨᓴᓕᓚᒃᓗᑕ ᑐᒃᑐᓂᒃ. ᓂᕆᐅᒃᑲᓗᐊᖅᐳᒍᑦ ᑖᓐᓇ ᐱᕚᓕᕈᑕᐅᓂᐊᕐᓂᖓᓂᒃ ᖃᓪᓚᑦᔪᓐᓇᖅᓯᓗᑕ ᐃᒪᐅᑉ ᖁᕐᓵᒍᑦ ᖃᓪᓚᑦᑕᖅᑎᓚᒃᓴᑦ ᐊᔾᔨᓕᐅᕈᑎᓄᑦ ᓯᕗᓂᒃᓴᑦᑎᓐᓂ ᖃᐅᔨᓴᕐᓂᕆᒃᑦᑕᙵᖅᑕᑦᑎᓐᓂᒃ.

- **ᒋᐅᕆ:** ᑎᑎᖅᑲᖁᑎᖃᖅᐱᑦ ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᔪᖅ ᓇᓄᐃᑦ ᐆᒪᔅᓲᔭᕈᓐᓇᕐᒪᖕᒫᑕ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ? ᐊᒃᑐᐃᕙᖕᒪᑕ ᐊᖑᓇᓱᒍᓯᖏᓐᓂᑦ ᐊᖑᓇᓱᒍᓯᕆᕙᒃᑕᖏᓐᓂᓪᓗ.

- **ᐋᓪᓚ**: ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦ ᖁᖓᓯᕈᖅᓯᓚᐅᙵᓐᓇᓕᖅᑐᑦ ᓇᓄᕐᓂᑦ ᐅᑭᐅᑦ ᖁᓕᑦ ᐅᖓᑖᓄᑦ. ᓱᓕ ᓇᓄᖅᑕᖅᐳᖅ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓂᒃ ᐊᕙᑎᓕᕆᔾᑯᑦ ᓯᓚᐅᓪᓗ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᑲᓇᑕᒥ ᒫᓂᒃᐸᓖᕐᔪᑎᒃ. ᖁᖓᓯᕈᖅ ᐊᕐᕋᒍᓄᑦ 1-3-ᓄᑦ ᑲᑦᑲᒃᔭᖅᑐᖅ. ᐅᖃᖃᑎᒋᓪᓗᒋᑦ ᖃᐅᔨᓴᖅᑎᐅᔪᑦ ᖁᖓᓯᕐᒥᐅᑕᐃᑦ, ᖃᐅᔨᒪᓇᓱᐊᖅᐸᒃᑐᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ. ᐱᔪᓇᕐᓂᓕᒫᒥᒍᑦ ᐱᓕᕆᓇᓱᐊᖃᑦᑕᖅᑐᑦ ᑕᒪᒃᑯᐊ ᖁᖓᓯᕈᐃᑦ ᐋᔭᖅᑕᐅᖃᓗᐊᕐᒪᖕᒫᑕ, ᑭᓯᐊᓂ ᓇᓄᐃᑦ ᐃᒪᕐᒦᑦᑐᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑎᓯᖏᓐᓂ ᐊᔪᕐᓇᖅᑎᑦᑎᕙᖕᒪᑕ. ᐊᕐᓇᑐᐊᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ. ᖃᓪᓚᑦᑎᐊᓕᓴᐅᓀᖅ ᓈᓄᒃᑕᐅᓚᐅᖅᑐᖅ ᖁᖓᓯᕈᖅᑕᐅᓪᓗᓂ ᑎᑭᕋᕐᔪᐊᕐᒥ; ᑕᐃᓐᓇ ᓇᓄᖅ 24-ᓂᒃ ᐅᑭᐅᖃᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᓈᓄᒃᓯᒪᓕᖅᑐᑦ 30-ᓂᒃ ᐅᑭᐅᖃᖅᑐᓂᒃ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓂᒃ, ᑭᓯᐊᓂᓕ ᑕᐃᓐᓇ ᓇᓄᖅ ᑕᐃᒪᐃᒐᔪᙱᓗᐊᕐᒪᑦ. ᖁᔭᓐᓇᒦᒃ ᑐᓴᖅᑎᑦᑎᓚᐅᕐᕕᑦ ᐃᓱᒫᓗᑎᓂᒃ ᖁᖓᓯᕈᓕᖅᓱᐃᓂᐅᑉ ᒥᒃᓵᓄᑦ ᖃᐅᔨᓴᖅᑎᓂᓪᓗ ᖁᖓᓯᕈᖅᓱᐃᓱᑦ ᓇᓄᓐᓂᑦ ᐋᓂᖅᓯᔪᒪᙱᒪᑕ ᑐᑭᓯᔪᒪᓪᓗᑎᓪᓗ ᖃᓄᖅ ᖁᖓᓯᕈᐃᑦ ᐊᒃᑐᐃᕙᒪᙱᖏᑦ ᓇᓄᕐᓂᑦ.

- **ᔫᓂ:** (ᐋᓪᓚ ᐅᑎᕐᓗᓂ ᖃᓪᓚᑦᕗᖓᑕ ᑐᑭᓕᐊᖓᔫᓴᒧᑦ) ᓇᓗᓇᐃᙶᖓᕕᒋᑦ ᕿᕐᓂᖅᑕᑦ ᑐᑭᓕᐊᖅᑐᑦ (ᓇᓄᐃᑦ ᑭᓪᓕᖏᑦ) ᑭᓲᕙ? ᑕᒪᓐᓇ ᑕᒃᓴᐅᑎᑦᑎᕞ ᓇᓃᒻᒪᙱᑦ ᖃᐅᔨᓴᕐᓂᖅ? ᐃᓛᓐᓂᒃᑯᑦ ᓇᓄᐃᑦ

ᓄᓇᖅᐸᓯᒃᑐᒦᓐᓂᖅᓴᐅᕙᖕᒪᑕ ᐊᒻᒪᓗ ᓯᔾᔭᒦᒐᑎᒃ ᓇᓄᕐᓂ ᑕᑯᙱᑐᐃᓐᓇᓇᐊᖃᕋᕕᑦ. ᖃᐅᔨᓴᖅᑐᖃᓚᐅᖅᐸ ᐱᙵᓐᓇᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᑉ, ᓂᒋᖅᐸᓯᐊᓂ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐊᒻᒪᓗ ᕿᑭᖅᑕᕐᒥᓐᓂ, ᐊᒻᒪᓗ ᑯᐸᐃᑉ ᓯᔾᔭᖓᓂ?

- **ᐊᓖᓴ**: ᕿᓂᖅᑕᖅ ᑐᑭᓕᐊᒃᑐᖅ ᓄᓇᙳᐊᑉ ᑭᒡᓕᒋᕙᐃᑦ ᓇᓅᖃᑎᒌᓄᑦ. ᐊᓯᖏᑦ ᑕᖅᓴᖏᑦ ᑐᑭᓕᐊᒃᑐᑦ ᑕᑯᑎᑦᑎᔪᑦ ᖃᖓᑕᕝᕙᖓᑕ ᑐᑭᓕᐊᖓᓂᖏᓐᓂ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᑕ ᑭᒡᓕᖓᓂ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᕈᑕᐅᓂᐊᖅᑐᑦ ᓇᓂᓐᓂᖏᑦ, ᑯᐸᐃᒃ ᓯᔾᔭᖓᓂ, ᓴᓂᑭᓗᐊᑉ ᖃᓂᒌᔭᓂᑐᐊᙳᙱᑦᑐᖅ. ᖁᔭᓐᓇᒦᒃ ᔫᓂ ᖃᐅᔨᑎᒃᑲᑦᑎᒍᑦ ᓯᓚ ᓇᓄᖅᑕᖃᒃᑲᓐᓂᕐᓂᖅᓴᖓᓂᒃ ᓄᓇᖅᐸᓯᖓᒥ.

- **ᓵᓕ:** ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᔪᖅ ᖁᙵᓯᕈᖅ ᐊᑐᖅᑕᐅᓲᙳᕙ? ᐅᑭᐅᓄᑦ ᐊᑕᐅᓯᕐᒥ ᐱᙵᓱᓄᑦ? ᐹᑕᓖᓄᑦ ᐊᐅᓚᑕᐅᕙᑦ?
- **ᐊᓖᓴ**: ᐄ, ᐃᒪᓐᓇᐅᓚᔪᖕᓂᖅᓴᑦ ᐊᑐᖅᑎᒋᔭᓐᓇᖅᑐᑦ ᐅᑭᐅᓕᒫᖅ ᓇᑉᐸᕐᓗ ᑭᓯᐊᓂᑦᑕᐅᖅ ᐅᑭᐅᓄᑦ ᐱᙵᓱᓄᑦ ᑎᑭᐅᑎᖃᑦᑕᕐᒥᔪᑦ.
- **ᓕᓯ:** ᖃᓇᑕᐅᔭᒃᑯᑦ ᐋᖅᑭᒃᑎᓯᒪᕙᑦ ᑲᑕᒍᓐᓇᖅᓱᑎᒃ?
- **ᐊᓖᓴ**: ᐄ ᑕᐃᒪᐃᑦᑐᑦ.
- **ᓵᓕ:** ᖃᖓᒃᑲᓐᓂᖅ ᐊᑖᑕᒐ ᓇᓄᓚᐅᖅᓯᒪᖕᒪᑦ ᖁᙵᓯᕈᖅᓯᒪᔪᒥᒃ (ᖃᖓᑲᓪᓛᓗᒃ). ᑐᓐᓂᖅᑯᑎᓕᐅᖅᑕᖓ ᖁᙵᓯᕈᖅ ᒪᕙᒪᒃᑯᓐᓄᑦ ᐃᓯᒪᓚᐅᕐᒪᑦ ᓱᓕ ᐊᐅᓚᓂᖅᑲᖅᓯᕐᓗᓂᐅᒃ.
- **ᐊᓖᓴ**: ᐃᓛᓐᓂᒃᑯᑦ ᑕᐃᒪᐃᓲᑦ! ᓴᓇᓯᒪᔫᓗᐊᑦ ᖃᖓᑦᑕᖅᑎᑕᐅᓯᒪᔪᒧᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᓂᒃ ᖃᐅᔨᕝᕙᐅᔭᓐᓇᖅᓱᑎᒃ, ᑭᓯᐊᓂᓗ ᐊᔪᙱᑦᑐᐃᓐᓇᐅᖃᑦᑕᙱᓪᓗᒪᑕ, ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᑯᑦ ᐃᓚᖏᑦ ᐱᑕᖃᖃᑦᑕᙱᑦᑐᑦ ᖃᓇᑕᐅᔭᕐᒦᑦᑐᑦ. ᖁᙵᓯᕈᕐᒥᒃ ᐃᑭᑕᐅᔭᖃᖅᐸᑦ, ᒥᓇᑎᔭᓐᓇᖅᑕᕗᑦ ᑐᓴᐅᒪᔭᒃᓴᓕᒫᑦ ᖁᙵᓯᕈᕐᒦᑦᑐᑦ ᑲᑉᐳᑎᓗᒋᑦ ᖃᓇᑕᐅᔭᕐᒧᑦ. ᐃᓄᐃᑦ ᐱᔪᒪᓐᓂᖅᐸᑕ, ᓇᓄᒃᑐᖃᓚᐅᕐᒪᑦ ᓴᓂᑭᓗᐊᕐᒥ ᑕᒫᓂ ᐊᕐᕌᒍᒥ ᑐᑭᓯᒋᐊᕈᑎᒃᓴᐅᔪᓂᒃ ᐱᓯᒪᒐᒪ ᐊᒻᒪᓗ ᓄᓇᙳᐊᒥᒃ ᓇᓅᒃᑲᔭᕐᒪᖔᑦ. ᔫᓇ ᑐᑭᓯᒋᐊᕐᕕᒃᓴᒃᑲᓐᓂᐅᖅᑯᖅᑐᖅ.

- **ᒨᓯᓯ:** ᖃᐃᓛᖅᐳᓯ ᑕᐃᒃᑯᓄᙵ ᐃᓱᒪᓕᐅᖅᑲᑦᑕᖅᑐᓄᑦ ᑰᑕᖏᑦ ᖃᓄᐃᑦᑑᓂᐊᕐᒪᖔᑕ, ᓇᓄᒐᓱᓐᓇᐅᑎᓪᓗᒍ. ᒪᒃᑯᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ, ᐱᕈᓃᖅᓯᒪᙱᑦᑐᑦ ᓇᓄᐃᑦ, ᐊᖑᓇᓱᒍᒪᒐᔭᖅᑐᒍᑦ, ᑭᓯᐊᓂ ᒫᓐᓇ ᐊᖑᓇᓱᒍᓇᕈᓐᓃᖅᑐᑦ. ᐃᓄᐃᑦ ᐃᓱᒪᓕᐅᕐᑯᔭᕗᑦ ᑕᐅᓄᖓ ᐊᒻᒪᓗ ᑐᓴᕐᓗᑎᒃ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ ᐊᖑᓇᓱᓐᓂᐅᑉ ᒥᒃᓴᓄᑦ ᓂᕆᐅᑦᑕᑦᑎᓐᓂᒃ. ᑭᓯᐊᓂ ᓇᓄᒍᓇᕐᓴᑦᑕ ᐃᓐᓇᕐᓂᒃ. ᑕᑯᔪᒪᔭᒃᑲ ᑕᐃᒃᑯᐊ ᐃᓄᐃᑦ ᑕᒪᐅᙵ ᑎᑭᓪᓗᑎᒃ ᑲᑎᒪᖅᑲᑎᖅᑲᓇᐊᖅᑐᖅᑐᑎᒃ ᐅᕙᑦᑎᓐᓂᒃ. ᑎᑎᕋᕐᓗᐊᖅᑐᒍᑦ ᐱᔪᓐᓇᖅᑎᑦᑎᓗᑕ ᐊᖑᓇᓱᒍᓇᕐᓗᑕ ᒪᒃᑯᓐᓂᖅᓴᓂᒃ ᓇᓄᕐᓂᒃ.
- ᖁᔭᓕᒐᔭᖅᐳᒍᑦ ᐊᒻᒪ ᑐᓴᕈᑦᑕ ᑖᒃᑯᐊ ᐱᓕᕆᔨᓪᓚᕆᑦ ᖃᐃᓂᐊᕐᒪᖔᑕ. ᓴᓂᑭᓗᐊᖅ ᕿᑭᖅᑕᕐᒥᑦ, ᒥᑎᕐᒥᐅᑕᐅᓚᑦᑕ ᐊᒻᒪᓗ ᐅᕙᑦᑎᓐᓂᑦ ᐱᒻᒪᓇᐅᓪᓗᑎᒃ. ᐊᒥᓲᓗᐊᕐᑎᓪᓗᒋᑦ ᓇᓄᐃᑦ ᓄᓇᒥ, ᒥᑏᑦ ᒪᓐᓂᕐᒥᑦ ᓂᖃᐅᓗᐊᖅᑲᑦᑕᕐᒪᑕ ᓇᓄᕐᓄᑦ. ᐊᒻᒪᓗ ᑕᒪᓐᓇ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᑯᔪᒪᕗᖓ ᐃᓄᐃᑦ ᑭᖑᓪᓕᖅᐹᒥ ᐃᓱᒪᓕᐅᖅᑲᑦᑕᖅᑐᑦ ᓇᓄᐃᑦ ᒥᒃᓵᓄᑦ.

20:05 – 20:10 ᓄᖅᑲᖓᓚᐅᑲᖓᖅ

ᖃᓄᖅ ᐅᓄᖅᑎᒋᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕐᓂᖅ/ᓯᓚᒥ ᐱᓕᕆᓂᖅ

- **ᔮᓇᑕᓐ ᑭᐅᓪᓗᓂ ᒨᓯᓯᐅᑉ ᐅᖃᐅᓯᕆᓚᐅᖅᑕᖏᓐᓄᑦ** ᑐᑭᓯᓇᖅᓯᑎᓪᓗᒍ ᑭᑭᑕᐅᓯᒪᓂᖏᓐᓂᒃ ᐊᑎᖅᑕᓛᓐᓂᑦ/ᒪᒃᑯᒃᑐᑦ ᓇᓐᓄᒃᑕᐅᔭᓐᓇᖅᑐᑦ.
- **ᔮᓇᑕᓐ:** ᑕᒪᓐᓇ ᐃᓕᖅᑯᓯᐅᕙᓪᓚᑦ ᐊᒻᒪᓗ ᐊᒥᓱᑲᓪᓚᐅᓪᓗᑎᒃ ᒪᓕᒃᑕᐅᔭᓇᐊᖅᑐᑦ ᐊᒻᒪᓗ ᒪᓕᒋᐊᓕᑦ, ᑭᓯᐊᓂᓚ ᑕᐃᒪᐃᑦᑐᖅ. ᐊᕝᕕᑐᖅᓯᒪᔪᓂ ᐊᐅᓚᑦᑎᔨ, ᓯᕝ ᒥᒃᑖᓄᓪ, ᐊᒻᒪᓗ CO ᐅᐃᓚᐊᒻ ᕙᓚᐅᕆᑎ, ᖃᐅᒃᐸᑦ ᐅᕙᓃᓐᓂᐊᕐᒪᑕ ᐊᒻᒪᓗ ᓇᓗᓇᐃᖅᓯᑦᑎᐊᕈᓇᕐᓂᖅᓴᐅᓪᓗᑎᒃ ᖃᓄᖅ ᑕᒪᒃᑯᐊ ᑭᑭᑕᐅᓯᒪᓂᕐᒧᑦ ᐃᓕᖅᑯᓯᐅᕙᓪᓚᑦ.

- ᑲᑎᒪᔨᑦ ᐊᓂᐊᓇᐃᒍᓱᖅᑰᔨᓚᐅᖅᑐᑦ ᑕᒪᓐᓇ ᐱᑕᖃᕐᓂᖔᓂᒃ ᖃᐅᔨᒪᙱᓐᓂᖏᑦ; ᔫᓇ ᐅᖃᓚᐅᖅᑐᖅ ᖃᐅᔨᒪᖕᖏᓐᓂᕋᖅᓱᓂᑦᑕᐅᖅ. ᐅᖃᖅᑲᐅᖕᒥᖕᒪᑦ ᐃᓱᒪᓕᐅᖅᑎ ᑲᑎᖃᑎᖃᕈᒪᓂᕋᖅᓱᑎᒃ, ᒥᓂᔅᑕ ᑖᓂᐊᓕ ᖃᕝᕕᒃ ᓴᓂᑭᓗᐊᕐᒥ ᒪᓕᒐᓕᐅᖅᑎ.
- **ᐊᖑᓇᓱᒃᑐᓕᕆᔾᒃᑯᑦ ᐅᖃᐅᓯᒃᓴᑐᐃᓐᓇᖏᑦ** ᔭᕝᒥᒃ ᑲᑎᖃᑎᖃᕈᒪᓚᐅᖅᐳᑦ ᐊᒻᒪ ᓇᓗᓇᐃᖅᓯᓗᑎᒃ ᓴᓂᑭᓗᐊᕐᒦᓐᓂᖖᓂ.

- **ᐊᓖᓴ**: ᑕᓯᐅᔭᕐᕙᐊᑉ ᓂᕆᖅᐸᓯᐊᓃᑦᑐᑦ ᓄᓇᓖᑦ, ᐃᓄᐃᑦ ᐅᖃᓚᐅᖅᐳᑦ ᐊᓄᓇᕐᕙᐊᑲᑕᓚᐅᕐᓂᖖᓂᒃ 2021−ᒥ ᓯᑯᓗ ᐊᔭᒃᑕᐅᓯᒪᓚᐅᖅᓱᓂ ᑕᓯᐅᔭᕐᕙᑉ ᓂᕆᖅᐸᓯᐊᓄᑦ, ᓇᓄᐃᑦ ᓈᓴᖅᑕᐅᓂᖅᓴᐅᓕᖅᓱᑎᒃ ᑕᓯᐅᔭᕐᕙᐊᑉ ᓂᕆᖅᐸᓯᐊᓃᑦᑐᑦ 2021−ᒥ. ᑲᑎᒪᔩᑦ ᖃᐅᔨᓚᐅᖅᐸᑦ ᓈᒻᒪᙱᓪᓕᐅᕈᑕᐅᕙᓂᒃ 2021−ᒥᑦ?
- **ᒨᓯᓯ:** ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᕆᙱᑎᒍᑦ, ᐅᖃᐅᓯᑐᐃᓐᓇᓇᙱᒃᑯᑦᑎᒍᑦ ᐊᒻᒪᓗ ᑎᑎᕋᖃᑦᑕᕈᑦᑎᒍᑦ ᖃᐅᔨᔭᕗᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓯᒪᔭᕗᑦ ᐊᔾᔨᒋᓚᐅᙱᓪᒪᒋᑦ, ᑎᑎᕋᖃᑦᑕᕋᔭᓚᐅᖅᑕᕗᑦ. ᐊᐃᑦᑖᖑᒐᓗᐊᖅ, ᑕᒪᓐᓇ ᐃᓄᐃᑦ ᐱᖅᑯᓯᕆᓪᒪᐅᒃ ᑎᑎᕋᕆᐊᒃᓴᖅ ᖃᕋᑕᕐᓄᑦ ᐊᒻᒪᓗ ᐸᐃᑉᐹᒃᑯᑦ ᑎᑎᕋᖅᑕᐅᒐᔪᙱᓐᖑᓂᒃ. ᓇᓗᓇᙱᑦᑐᖅ ᑎᑎᖅᑲᕐᑯᑎᖃᕐᓂᕈᑦᑕ ᐊᖑᓇᓱᒃᑎᐅᓪᓗᑕ, ᑲᑐᔾᔨᒐᔭᖅᐳᑦ ᑭᓯᐊᓂ ᐊᖑᓇᓱᒃᑎᐅᓪᓗᑕ, ᐊᕐᕌᒍᑕᓖᑦ ᐊᒻᒪ ᐊᕐᕌᒍᑕᓖᑦ ᐊᔾᔨᒌᑉᐸᙱᓪᒪᑕ. ᐊᓯᔾᔨᖅᐸᓪᓕᐊᖏᓐᓇᐅᔭᕐᒪᑕ ᐊᕐᕌᒍᑕᓖᑦ.
- ᓄᓇᓕᖕᒥᐅᑕᐅᓪᓗᓂ ᐊᕙᑎᑦᑎᓐᓄᑦ ᑕᑯᕚᒃᑐᓂ ᓯᓚᐅᑉ ᐊᓯᔾᔨᐸᓪᓕᐊᓂᖓᓂᒃ, ᐃᓄᐃᑦ ᖃᐅᔨᒪᓂᑐᖃᕆᙱᑎᒍᑦ, ᖁᑦᑎᓂᖅᐹᒧᖅᐸᔪᔪᑦ ᖃᓄᑎᒋ ᓯᑯᐃᙱᓐᓃᔭᖅᑎᒋᓇᔭᕐᒪᖔᑦ. ᑕᒪᓐᓇ ᐊᔪᕐᓇᕐᒪᑦ ᐅᓪᓗᒥ, ᐊᓯᔾᔨᖅᑐᒻᒪᕆᐅᓂᕐᒪᑦ. ᒫᓐᓇ ᖃᐅᔨᔪᓐᓇᕈᓐᓃᖅᑐᒍᑦ ᓯᑯ ᖃᓄᐃᓛᕐᒪᖔᑦ. ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᐃᖏᕐᕋᑲᑕᓗᐊᒧᑦ ᑕᐃᒪᐃᓪᒪᑦ ᓇᓗᓇᖅᐳᖅ ᖃᓄᖅ ᓯᑯ ᐃᖏᕐᕋᖕᒪᖔᑦ ᐅᕝᕙᓘᓐᓃᑦ ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᓂᐊᕐᒪᖔᑦ.
- **ᒋᐅᓇ:** ᐊᕐᕌᒍᑕᖃᓚᐅᕐᒥᔪᖅ, ᐃᖅᑲᐅᒪᙱᑦᑐᖓ ᓇᓂᐊᖑᓚᐅᕐᒪᖔᑦ ᐊᕐᕌᒍ, ᐊᒥᓱᐊᓗᖕᓂᒃ ᓇᓄᑦᑐᖃᓚᐅᖅᓯᒪᔪᖅ ᐊᐅᔭᒃᑯᑦ. ᓇᓄᐃᑦ ᓯᑯᐃᕈᔾᔭᐅᓚᐅᖅᑐᑦ ᐊᔪᖅᓯᐅᖅᑲᓪᓗᑎᒡᓗ ᕿᑭᖅᑕᓄᑦ. ᑕᐅᓄᙵᓚᐅᖅᑐᒍᑦ ᐊᒻᒪᓗ ᐊᒥᐊᒃᑯᖃᓗᐊᕋᓂ ᑲᑎᖅᓱᕝᓴᓂᒃ. ᐊᒥᓱᓂᒃ ᓇᓄᕐᓂᒃ ᓱᕋᒃᑎᕆᔪᖃᖃᑦᑕᓚᐅᖅᐳᖅ ᒥᑏᑦ ᐃᕙᕝᕕᖏᓐᓂᒃ.
- **ᐱᓕᕆᖃᑎᒌᑦ:** 2021-ᖑᑎᓪᓗᒍ.
- **ᓵᓕ:** ᒨᓯᓯᐸ ᐅᖃᖅᑲᐅᔭᖓ ᐅᐊᑦᑎᐊᒃᑳᓂᖅ, ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᕐᒋᑦ ᐱᓕᕆᔾᔪᓯᖅ ᐊᔾᔨᒌᙱᑕᖓᓂᒃ ᐱᓕᕆᔾᔪᓯᖃᕐᒪᑦ. ᓯᑯ ᓵᑦᑐᑯᔫᓕᕐᒪᑦ. ᑕᒪᓐᓇ ᐊᓯᔾᔨᕐᓂᖅᐸᐅᖃᑕᐅᖕᒪᑦ ᑕᑯᕝᑕᑦᑎᓐᓂ ᐊᖑᓇᓱᒃᑎᐅᓪᓗᑕ. ᐊᒻᒪᓗ ᓱᑲᑦᑐᒥᒃ ᓴᖅᑭᖅᑐᓂ.
- **ᔫᓂ:** 2021−ᒥ ᐊᐅᔭᒃᑯᑦ, ᐃᓚᐅᓚᐅᖅᐳᖓ ᒥᑎᕐᓂᒃ ᐃᕙᕝᕕᖏᓐᓂᒃ ᖃᐅᔨᓴᖅᑐᓂᒃ ᓴᓂᑭᓗᐊᐸ ᕿᑭᖅᑕᖓᑕ ᓂᕆᐊᓂ ᐊᒻᒪ Sleeper ᕿᑭᖅᑕᓂᒃ. ᑕᒪᐃᓐᓂ ᐃᕙᕝᕕᖃᑦᑎᐊᓚᐅᙱᑦᑐᖅ. ᓇᓗᓇᙱᑦᑐᖃᐅᑦᑎᐊᓚᐅᖅᐳᖅ ᐃᕙᕝᕕᖏᑦ, ᑭᓯᐊᓂ ᑕᐃᒃᑯᐊᓂᓖᖅ ᐃᕙᕝᕖᑦ ᒪᓂᖃᓚᐅᙱᓪᒪᑕ ᓇᓄᕐᓄᑦ ᐸᓪᕕᔮᖅᑕᐅᓂᑯᑦ. ᐅᑭᐊᒃᓵᖅ, ᑭᓇᑭᐊᖅ ᓯᕿᓂᕐᒥᐅᑕᖅ, ᑯᓇᒃᓴᔭᐅᓪᓗᓂ, ᐊᐱᕆᓚᐅᕐᒪᑦ ᖃᓄᐃᓚᐅᕐᒪᖔᑦ ᐊᐅᔭᕗᑦ ᐱᓪᓗᒍ ᐊᒥᓱᑦ ᒥᑏᑦ ᓅᓚᐅᕐᒪᑕ ᓯᕿᓂᕐᒧᑦ ᐊᒻᒪ ᐃᕙᓚᐅᖅᓱᑎᒃ ᑕᐃᑲᓂ ᐊᕐᕌᒍᒥ.

- **ᓵᓕ:** ᑕᐃᒃᑯᐊ ᖃᐅᔨᓴᕈᑕᐅᓚᐅᖅᑐᑦ ᐅᖃᐅᓯᕆᕋᑖᖅᑕᑎᑦ (ᑕᓯᐅᔭᕐᕙᐊᑉ ᐱᖓᓐᓇᖅᐸᓯᐊᓂ ᓂᕆᖅᐸᓯᐊᓂᓗ), ᐃᓄᐃᑦ ᕿᑭᖅᑖᓗᖕᒥ ᓇᖕᒥᓂᖅ ᖃᐅᔨᓴᖅᓯᒪᕙᑦ, ᐊᒥᒐᓂᖕᒋᓐᓂᒃ ᖃᐅᔨᓴᕐᓂᖃᖅᐸᑦ, ᓇᐅᒃᑲᖅᑕᐅᑎᓂᑦ, ᐊᒻᒪᓗ ᑕᒪᒃᑯᐊᕈᓘᔭᕐᓄᑦ? ᐊᒻᒪ ᑯᐸᐃᒃ ᑕᕐᕋᖓᓂ? ᖃᐅᔨᒪᔪᒍᑦᑕᐅᖅ ᓇᓄᖃᑦᑕᕐᒪᑕ. ᑐᑭᓯᒋᐊᕈᑎᒃᓴᖃᖅᑐᒃᓴᐅᕗᑦ ᓇᐅᒃᑲᖅᑕᐅᖃᑦᑕᕐᒪᖔᑕ ᐊᒻᒪ ᐊᖑᔭᐅᕙᖕᒪᖔᑕ.
- **ᐊᓖᓴ**: ᐋ, ᐃᖏᕐᕋᔩᖑᔪᒍᑦ/ᓂᑭᑦᑖᖅᓱᑕ ᖃᐅᔨᓴᖅᓱᑕ, 12-ᓂᒃ ᐊᐅᓚᑦᑎᔭᕆᐊᖃᕐᓂᑦᑎᓐᓄᑦ. ᑖᓃ ᐊᕐᕌᒍᒥ ᐸᕐᓇᐃᓯᒪᔪᒍᑦ ᖃᐅᔨᓵᕐᓗᑕ ᓯᑯᓰᓚᕐᒥᐅᑦ ᑕᕆᐅᖓᓂ ᐱᖃᓯᐅᔾᔨᔪᖅ ᑯᐸᐃᖕᒥ ᐊᒻᒪᓗ ᕿᑭᖅᑖᓗᖕᒥ ᐊᒻᒪᓗ ᐅᐊᖕᓇᖓᓂ ᑭᕙᓪᓕᕐᒥ. ᑎᒥᖏᓐᓂᑦ ᐲᖅᓯᓇᓱᐊᕋᑦᑕ ᐊᒻᒪᓗ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᓱᑕ. ᐃᖅᑲᓇᐃᔭᖃᑎᕗᑦ ᑯᐸᐃᖕᒥ ᑕᐅᑐᓪᓗᐊᑕᕐᓂᐊᖅᑐᑦ ᐃᓚᖓᓂ ᖃᐅᔨᓴᕐᓂᕐᒥ ᑖᓃ (ᑎᒃᑯᐊᖅᓯᓗᑎᒃ ᑯᐸᐃᒃ ᓯᓪᔭᖓᓄᑦ). ᐊᐅᓚᖅᑎᑦᑎᔪᒪᔪᒍᑦ ᐊᖑᓇᓱᒃᑕᐅᓂᑯᓂᒃ ᖃᐅᔨᓴᕋᒃᓴᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓗᑕ ᓇᐅᔮᑦ ᐲᖅᓯᕖᐅᓯᒪᖕᒪᖔᑕ ᓇᐅᒃᑲᖅᑕᐅᑎᓂᑦ ᑕᓯᐅᔭᕐᕙᐊᑉ ᓂᕆᖅᐸᓯᐊᓂ ᓇᐅᒃᑲᖅᑕᐅᒻᒪᑦ ᓯᑯᓯᓛᕐᒥᐅᑦ ᑕᕆᐅᖓᓂᑦ. ᖃᐅᔨᓴᖅᑎᒻᒪᕇᑦ

ᑐᓴᒃᓴᕐᒥᑦ ᐱᓯᒪᔭᕗᑦ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖅᐸᓯᐊᓂ ᖃᐅᔨᓴᖅᑕᐅᓂᖅᐹᖑᕐᒪᑕ. ᑕᐃᒪ ᐊᔾᔨᒌᓂᒃ ᑐᓴᒃᓴᖃᙱᑐᐃᓐᓇᕆᐊᖃᖅᑐᒍᑦ ᐊᓯᖏᓐᓄᑦ.

- ᐱᒪᕆᐊᓘᔪᖅ ᐱᑎᑦᑎᔭᕆᐊᒃᓴᖅ ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᖃᐅᔨᓴᒃᓴᓂᒃ ᐆᒪᔪᕐᓂᐊᖅᑎᒃᑯᑦ ᑎᑎᕋᕐᕕᖕᓚᓄᑦ. ᐃᑲᔪᖅᑕᑦᑕᖅᑐᖅ ᐅᕙᑦᑎᓐᓂᑦ ᑭᐅᓂᕐᒥᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᐱᖅᑯᑎᓂᑦ ᑖᔅᓯᒪᙵᑦ, ᑭᓯᐊᓂ ᐱᒪᕆᐅᖕᒥᔪᖅ ᐅᕙᑦᑎᓐᓄᑦ ᓴᐳᔾᔨᔨᓐᓇᓂᕐᒥᑦ ᓄᓇᕗᒻᒥ ᐊᖑᓇᓱᖕᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᑐᓂᑦ ᓯᓚᕐᔪᐊᕐᒥᑦ. ᑲᑎᒪᖃᑦᑕᖅᓯᒪᕗᖓ ᐊᓯᖏᑦ ᓄᓇᓕᒃᔪᐊᑦ ᐊᖑᓇᓱᒍᓐᓇᑎᑦᑎᕙᙱᓪᓗᑕ ᐊᓪᓚ ᓈᓚᒃᓴᖅᐸᙱᕐᓱᑎᒃ ᓄᓇᕗᒻᒥ ᐊᐅᓚᑦᑎᔾᔪᓯᕐᓂᒃ, ᑭᓯᐊᓂ ᐃᑲᔫᑎᖃᖅᑐᒪᓇᐅᕗᖅ ᑎᑎᖅᑲᖅᑯᑎᖃᖅᑎᓪᓗᑕ ᐊᖑᓇᓱᖕᓂᐅᔪᓂᑦ ᓄᓇᓕᖕᓂᑦ ᐊᓪᓚ ᑐᓂᓯᕚᖕᕙᑦ ᖃᐅᔨᓴᒃᓴᓂᒃ ᓴᐳᔭᐅᖁᔨᓗᑎᑦ ᓄᓇᕗᒻᒥ ᐊᐅᓚᑦᑎᔾᔪᓯᑦ. ᑕᐃᒪᐃᓪᒪᑦ ᖁᔭᓐᓇᒦᖅᑕᐃᓐᓇᕆᔪᒪᕙᒃᑲ ᐊᖑᓇᓯᑦᑎᓕᒫᑦ ᑕᒪᒃᑯᓂᖓ ᖃᐅᔨᓴᖅᑎᑦᑎᓚᐅᕐᒪᑕ ᐱᓪᓗᒍ ᐃᑲᔫᑎᖃᕐᒪᑦ ᐊᖑᓇᓱᖕᓂᒃᑯᑦ.

- **ᓵᓕ:** ᐃᖅᑲᐅᒪᔪᖓ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᓚᐅᕐᒪᑕ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᕐᓂᖏᓐᓄᑦ ᐊᐅᔭᖅ. ᖃᐅᔨᒪᐸᓗᐸᑦ ᖃᖓ ᑭᖑᓪᓕᕐᒥ ᖃᐅᔨᓴᕐᓂᖃᓛᕐᒪᖔᑕ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ?
- **ᐊᓖᓴ**: ᐊᕐᖂᒍᓪᓚᑦᑖᒥᒃ ᓇᓗᓇᐃᖅᓯᓯᒪᙱᑦᑐᒍᑦ, ᑭᓯᐊᓂ ᐅᑭᐅᑦ ᑕᓪᓕᒪᑦ ᐊᓂᒍᕚᖓᑕ ᑕᐃᒪᐃᖅᑲᑦᑕᖅᓯᒪᓕᖅᑐᒍᑦ. ᒪᓕᒃᐸᖅ ᖃᔅᓯᓂᒃ ᕿᓇᐅᔭᓂᒃ ᐊᑐᐃᓐᓇᖅᑕᖃᕐᒪᖔᑦ ᐱᔾᔪᑎᒋᓗᒍ ᖃᐅᔨᓴᕐᓂᖅ ᐊᑭᑐᔪᒻᒪᕆᐅᓂᖓᓄᑦ. 2026−ᒥᖃᐃ ᒪᓕᒃᑯᑦᑎᒍᑦ.
- ᖃᖓᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᕆᐊᖃᙱᑦᑐᖅ. ᖃᐅᔨᓴᕈᓐᓇᑐᐃᓐᓇᕆᐊᖃᖅᑐᒍᑦ ᑎᒥᖏᓐᓂᑦ ᑕᒪᓐᓇ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑐᓴᕈᒪᒃᐸᑕ. ᐲᖅᓯᔪᖃᖃᑦᑕᖅᑎᓪᓗᒍ ᓇᐅᒃᑲᖅᑕᐅᒥᒃ, ᑕᑯᔪᓐᓇᕋᔭᖅᑐᒍᑦ ᐊᐅᓚᔾᔭᒃᐸᓪᓕᐊᔪᓂᒃ ᑎᑎᕋᖅᓯᒪᔪᓂᒃ, ᑭᓯᐊᓂ ᓯᕗᓂᒃᓴᑦᑎᓐᓂ ᐅᑎᕈᓐᓇᖅᓯᕙᖅᑐᒍᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᑐᓴᕋᓱᒃᑭᓗᒋᓪᓗ ᐅᖃᐅᓯᕆᓗᒋᓪᓗ ᐱᓕᕆᐊᖑᕕᒡᒍᓇᖅᑐᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᑲᔪᖅᓱᖅᑕᖏᑦ.

- **ᐃᓚᐃ:** ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦ ᐱᓕᕆᖃᑎᖃᖅᐸᒃᐸᑦ ᐃᕗᒃᑯᓐᓂ, ᐊᒻᒪᓗ ᐋᓐᓂᐅᕆᐅᒥᒃ ᐊᒻᒪᓗ ᓄᓇᕕᖕᒥ ᒐᕙᒪᖏᓐᓂᒃ ᖃᐅᔨᓴᖅᑎᓪᓗᒋᑦ?
- **ᐊᓖᓴ**: ᑭᖑᓂᑦᑎᓐᓂ, ᕿᓇᐅᔭᖃᖅᑎᑦᑎᖃᑦᑕᖅᓯᒪᕗᒍᑦ ᐊᓯᖏᓐᓄᑦ ᒐᕙᒪᐅᔪᓄᑦ ᖃᐅᔨᓴᕐᓗᓯᑦ. ᕿᓇᐅᔭᖅᑎᑦᑎᓯᒪᔪᒍᑦ ᐋᓐᓂᐅᕆᐅᒃᑯᑦ ᐊᒻᒪᓗ ᕿᑭᖅᑖᓗᒃ ᑯᐊᐳᐃᓴᓐ ᒐᕙᒪᖏᓐᓂᒃ ᓯᕗᒃᑲᖅᑎᐅᓂᐊᕐᓗᑕ ᑕᓯᐅᔭᕐᔪᐊᑉ ᓂᒋᖅᐸᓯᐊᓂ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᒻᒪᓗ ᐱᓕᕆᐊᖃᕐᓗᑕ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖓᓐᓇᖅᐸᓯᐊᓂ. ᓱᓕ ᐃᑲᔪᖅᑕᐅᔪᕆᐊᖃᖅᑕ ᐊᓯᖏᓐᓂᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓂᒃ ᐊᓪᓚ ᐱᔪᓐᓇᐅᑎᑖᖅᑎᑦᑎᓗᑕ, ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᐱᓕᕆᖃᑎᒌᒃᑕᖅᐳᒍᑦ. ᑖᒃᑯᐊ ᒐᕙᒪᒃᑯᑦ ᐃᑲᔪᖃᑦᑕᕐᒪᑕ ᖃᖓᑕᔫᒃᑯᑦ.
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦ ᓱᓕ ᓵᓂᑭᓗᐊᕐᒨᖅᓴᖅᑐᑦ ᐅᓂᒃᑲᓇᐊᖅᑐᕐᓗᑎᒃ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᕙᒃᑐᓂᒃ ᐊᒻᒪᓗ ᑐᓂᓯᓗᑎᒃ ᐱᔪᒪᔭᖏᓐᓂᒃ ᐆᒪᔪᕐᓂᐊᑎᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐋᓐᓂᐅᕆᐅᒃᑯᑦ ᒐᕙᒪᒃᑯᖏᓐᓄᑦ, ᑭᓯᐊᓂᓕ ᐅᕙᒍᑦ ᖃᐅᔨᓴᖅᑕᖃᖃᑦᑕᖏᓐᓇᑦᑕ.

ᑲᑎᕝᕙᒋᑦ ᖃᔅᓯᕌᕈᓐᓇᖅᑎᑕᐅᓂᖏᑦ ᐅᖃᖃᑎᒌᖕᓂᖅ

- **ᐃᓚᐃ:** ᑐᒃᑐᐃᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᓂᖏᓐᓄᑦ, ᓴᓂᑭᓗᐊᒥᐅᑦ ᑐᒃᓯᕋᕈᓐᓇᖅᐸᑦ ᐅᓄᖅᓯᑎᐊᖅᑕᐅᖁᔨᓗᑎᒃ ᑲᑎᕝᕙᒋᑦ ᖃᔅᓯᕌᕈᓐᓇᖅᑎᑕᐅᖏᓐᓂᒃ?
- **ᐊᓖᓴ**: ᑕᒪᓐᓇ ᓯᕗᒧᐊᒍᒥᓇᖅᑐᖅ ᑭᖑᓪᓕᕐᒥ ᑕᑯᓐᓇᓚᒃᓴᒃᑲᓐᓄᑦ. ᒫᓐᓇᐅᔪᖅ, ᓯᕗᓪᓕᖅᐹᖅᓯᐅᑎᕗᑦ ᑲᑎᕝᕙᒋᑦ ᖃᔅᓯᕌᕈᓐᓇᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᐅᖁᔨᔪᒧᑦ ᑲᑎᕝᕙᒋᑦ ᖃᔅᓯᕌᕈᓐᓇᖅᑎᑕᐅᓂᖓᓂᒃ, ᑭᓯᐊᓂᓗᑦᑕᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐱᔪᒪᒃᐸᑕ, ᑕᒪᓐᓇ ᐅᖃᐅᓯᕆᓇᕙᖅ ᑎᑎᖅᓯᒪᓕᖅᑎᑦᑐᒍᓂᒋᓗ ᑎᑎᖅᓯᒪᔭᑦᑎᓐᓂ.

- **ᒨᓯᓯ:** ᐱᒋᐊᕈᑎᖃᕈᒪᔪᖓ ᐅᓂᒃᑳᕐᓗᖓ. ᑕᐃᔅᓯᒪᓂᐊᓗᒃ ᐅᕙᖓᓗ ᐊᖓᔪᖓᓗ ᓇᑦᑎᖁᓯᕆᐊᓚᐅᖅᓯᒪᓚᖓᓄᒃ ᐊᒻᒪᓗ ᖁᓕᒥᒎᓕᖓᓄᑦ ᐋᐳᑎᓚᐅᖅᓯᒪᓕᕐᓗᑕ. ᑕᐃᓐᓇ ᖁᓕᒥᒎᓕᒃ ᑕᖅᑎᑦᑎᑦᑎᓚᓇᓚᐅᕐᒪᑦ ᓇᓄᕐᒥᒃ ᐃᑲᖅᑲᓚᐃᖅᓯᓐᓂᒃ ᒪᓕᒃᓴᖅᓯᓂᒧᖓᑦ ᐊᒻᒪᓗ ᓇᓄᖅ ᕿᒦᔾᔪᓇᐃᓕᓕᓚᐅᕐᒪᑦ ᑕᖅᑯᐊᕐᓂᒧᖓᑦ. ᑖᓇ ᓇᓄᖅ ᓯᓂᓕᖅᑎᑕᓚᐅᖅᑐᖅ. ᐃᓄᐃᑦ ᖃᐅᔨᒪᔾᔪᑎᖏᑦ ᐅᖃᐅᔾᔨᕐᒪᑕ ᐅᕙᑦᑎᓐᓂᒃ ᑭᓯᑐᐃᓐᓇᑦᑎᐊᕐᓂᒃ ᓂᕆᔪᑎᓂᒃ

ᓇᒡᓕᒃᕉᖅᑎᑦᑎᕆᐊᖃᙱᓐᓇᑦᑕ, ᑕᖅᑎᑦᑎᐊᕐᓗᒋᑦ ᑕᖃᓗᐊᕐᓂᑯᒧᓪᓗ ᓂᐅᕈᒪᓕᑦᑎᐊᕐᓗᑎᒃ. ᑕᐃᒃᑯᐊ ᐃᓄᐃᑦ ᑎᒥᙱᓐᓂᑦ ᐲᖅᓯᕚᒃᑐᑦ ᓇᐅᒃᑲᖅᑕᐅᒻᒧᑦ, ᓇᓄᕐᒥᑦ ᒪᓕᒃᓴᓗᐊᕐᓂᑯᒧᑦ ᐊᓂᖅᓵᕈᓐᓇᐃᓕᕚᖕᒪᑖ?

- **ᐊᓖᓴ**: ᐊᑯᓂᐅᓂᕆᓚᕝᑕᖓ ᑎᒥᙱᓐᓂᑦ ᐲᖅᓯᓇᓯᐊᕐᓂᖅ ᓇᐅᒃᑲᖅᑕᐅᒻᒧᑦ ᐊᒻᒪᓗ 1-ᒥᓂᑦ ᑐᖕᓕᓂᑦ. ᑕᒪᓐᓇ ᕿᓚᒥᑯᓘᑎᓪᓗᒍ, ᓇᓄᕐᒧᑦ ᐊᒃᑐᖅᑕᐅᙱᓐᓂᖅᓴᐅᖕᒪᑦ. ᐃᓕᑕᖅᓯᕗᒧᑦ ᓇᓄᕐᒥᒃ ᐸᒡᕕᓴᐃᓂᖅ ᐸᒡᕕᓴᐃᓂᖕᖓᓂᒃ/ᓈᒻᒪᙱᑦᑐᒻᒪᓇᐅᓂᖕᖓᓂᒃ ᓇᓄᕐᒧᑦ. ᑭᒡᓕᖅᑎᑦᑎᖃᑦᑕᓇᐊᖅᑐᒧᑦ ᓇᓄᕐᓂᒃ ᖃᐅᔨᓴᖅᑎᓪᓗᑕ ᐊᒃᓱᕈᖅᑎᑕᐅᓗᐊᕐᑯᓇᒋᑦ. ᒪᓕᒐᖅᑕᖃᐅᖅᑐᖅ ᖃᓄᖅ ᑎᒥᙱᓐᓂᑦ ᐲᖅᓯᖃᑦᑕᕐᒪᖔᑦᑕ ᓇᐅᒃᑲᖅᑕᐅᒻᒧᑦ. ᐃᓚᖏᑦ ᒪᓕᒐᐃᑦ ᐆᓇᕐᓂᕐᒧᑦ ᑭᐱᔭᐅᖃᑦᑕᖅᑐᑦ (ᓇᓄᐃᑦ ᐆᓇᓗᐊᖅᑎᓪᓗᒍ ᐱᑦᑕᐃᓕᓗᒋᑦ), ᓇᓄᕐᒥᒃ ᐅᖅᑰᓯᓗᐊᕐᑯᔾᔭᙱᑦᑐᒧᑦ. ᐃᖅᓯᓇᖅᑐᓂᒃ ᓇᓄᕐᓂᒃ ᒪᓂᕋᐅᙱᑦᑐᑏᑎᑦᑎᕙᙱᑦᑐᒧᑦ ᑲᑦᑯᓇᒋᑦ ᐊᒻᒪᓗ ᐃᒻᒥᓂᒃ ᐋᓂᖃᑦᑕᕐᑯᓇᒋᑦ ᐅᔭᖅᑲᓄᑦ. ᐊᒻᒪ ᐊᓈᓇᐅᔪᑦ ᐊᑎᖅᑕᓛᖅᑐᒥᓗ ᐊᕕᒍᒪᙱᒻᒥᒪᓗᒋᑦ. ᑕᐃᒪᓕ ᑕᑯᓯᒪᔭᒃᑯᑦ, ᐲᖅᓯᓂᖅ ᓇᐅᒃᑲᖅᑕᐅᑎᒥᒃ ᐸᕝᕕᓇᙱᓐᓂᖅᓴᐅᖕᒪᑦ ᐋᙳᓚᔾᔪᓇᖅᑐᒥᒃ ᐊᒻᒪᓗ ᖁᖕᓚᓯᕈᖅᓯᓂᕐᒥᒃ ᓇᓄᕐᒥᒃ.
- ᖃᐅᔨᒪᙱᓚᖓ ᖃᐅᔨᓴᖅᑎᓂᑦ ᑕᒪᐅᙵᖅᑳᖃᑦᑕᓚᐅᖅᑐᓂᑦ, ᑭᓯᐊᓂ ᖃᐅᔨᓴᖅᑏᑦ ᖃᐅᔨᒪᔭᒃᑲ ᐊᒃᑐᐃᓂᕐᓗᕆᓯᐊᖅᐸᖕᒪᑕ ᓇᓄᕐᓄᑦ. ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᑎᒥᙱᓐᓂᑦ ᐲᖅᓯᓂᕐᒧᑦ ᐱᓕᕆᐊᖅ ᓯᑯᓯᕈᕐᒥᐅᑦ ᑕᕆᐅᖕᒐᓂ, ᐊᒥᓱᓂᒃ ᐅᔮᖅᓱᕋᓱᐊᖅᑐᒧᑦ. ᖃᖕᒐᑦᓱᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᒪᓕᕆᐅᕈᙳᖕᒪᑦ ᖃᐅᔨᓴᕐᓂᒥᑦ, ᓇᐅᒃᑲᖅᑐᐃᓐᓇᕐᓂᐊᖅᐸᕗᑦ ᑐᐊᕝᕕᕐᓗᑕ ᐸᒡᕕᓇᙱᓪᓗᑎᓪᓗ. ᓇᐅᒃᑲᖅᑐᐃᓐᓇᖃᑦᑕᓲᖑᑎᑦᑐᒧᑦ ᐊᑦᑕᓇᖅᑐᒦᑎᓪᓗᑕ. ᐊᒻᒪᓗ ᓇᐅᒃᑲᖅᑕᐅᑎᒃᑯᑦ ᓇᐅᒃᑲᖅᑕᐅᑎᖃᑦᑕᓲᖑᑎᑕᕗᑦ ᓄᓇᓖᑦ ᖃᓂᒋᔭᓂᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᐃᑲᔪᖅᓯᒪᒥᖏᓐᓄᑦ.
- **ᒨᓯᓯ:** ᐅᖃᖅᑲᐅᒐᒪ ᑕᐃᒪ, ᐅᖃᐅᓯᒃᑯᑦ ᐅᓂᒃᑳᑦ ᑎᑎᕋᖅᑕᐅᕙᙱᓚᑕ. ᐊᖓᔪᖅᑳᒃᑲ ᐊᒻᒪᓗ ᐃᑦᑐᒃᑯᒃᑲ ᐅᖃᖃᑦᑕᓚᐅᕐᒪᑕ ᓇᓄᕐᓂᒃ ᑕᖅᓴᐃᖃᑦᑕᙱᓚᑕ ᐊᒻᒪᓗ ᑕᖅᑎᑕᐅᕙᙱᓚᑕ. ᐅᓪᓗᒥ ᖃᒧᑕᐅᔭᖃᖅᑕᑦ ᐊᒻᒪᓗ ᐊᖑᒻᒪᑎᕙᓐᓇᖅᓱᑕ ᓇᓄᕐᒧᑦ ᓯᑲᓐᓂᖕᓴᒧᑦ ᐊᒻᒪᓗ ᐊᖑᓗᒍ ᕿᒪᒃᓯᒋᐊᓚᐅᙱᓐᓂᖓᓂ. ᓇᓄᓚᓯᐊᕐᓂᖅ ᐃᓇᑐᖃᖅᑕᕐᓄᑦ ᖃᐅᔨᑎᖅᑐᓂ, ᐃᓇᑐᖃᐅᓂᖅᓴᑦ ᓂᕆᖃᑦᑕᙱᓚᑕ ᓇᓄᕐᒥᑦ ᕿᒦᓂᖁᓗᐊᕌᓗᖕᒥᑦ, ᑎᐱᖓ ᐊᓯᐅᔨᖃᑦᑕᕐᓂᖓᓄᑦ ᓇᓄᖅᓯᓐᓃᑦᑐᓐᓈᖅᓱᓂ. ᐊᖑᓇᓱᒃᐱᑦ ᖃᐅᔨᒪᖓᑕ ᓇᓄᕐᓄᑲᐅᖅᑐᓇᒃᓴᖅ ᓂᕆᑎᑦᑎᓂᐊᕐᒪᑕ ᐃᓄᖕᓂᑦ.

- **ᐊᓖᓴ**: ᑐᒃᓯᕋᓚᐅᕋᑦᑕ ᑖᒃᑯᐊ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᓱᕋᖁᒥᖕᖑᓗᒋᑦ ᑖᒃᑯᐊ ᒪᕐᕉᒃ ᓇᓄᐃᑦ ᐃᖕᒋᕋᓂᖕᒋᑦ ᒪᓕᒃᓱᓗᒋᑦ, ᐃᓱᒪᒋᔭᐅᓪᓗᑎᒃ ᖃᔾᔭᐅᓂᖕᒋᑦ ᑕᓯᐅᔭᕐᔪᐊᖑᒋᑦᑐᓕᖓᑦ ᓱᕋᓗᐊᖅᓯᒪᙱᑦᑐᐃᓐᓇᐅᓗᑎᒃ. ᑐᓴᖅᑑᔭᕋᒪ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓂ ᐃᓱᒪᓘᑎᑕᖅᑲᓪᓚᑦ ᐊᒥᓲᓂᖕᒋᓐᓂᑦ ᓇᓄᐃᑦ ᐊᒃᑐᐃᓂᖃᖅᑲᕐᒪᑕ ᖃᓯᐅᓂᖕᒋᓐᓂᑦ ᒥᑏᑦ. ᐃᒪᖄ ᓄᓇᓕᖕᓂ ᐆᒪᔪᕐᓂᐊᑎᒃᑯᑦ ᑲᑎᒪᔨᖕᒋᑦ ᐊᒥᓯᕈᓐᓇᐊᕐᓂᖅᓴᓪᓚᔭᖅᐸᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᑕᒪᓐᓇ ᐊᒃᑐᐃᓂᖅᑲᕐᓂᖓᓄ ᒥᑎᕐᓄᑦ? ᑕᒪᓐᓇ ᓱᓕᕚ?
- **ᐃᓚᐃ:** ᑕᒪᓐᓇ ᐅᖃᐅᓯᓇᑐᐃᓐᓇᖁᑖᖅᑕᖁᓗᐊᖑ, ᑭᓯᐊᓂᓕ ᑕᒪᓐᓇ ᐅᖃᐅᓯᓇᓂᐊᖅᑕᕗᑦ ᐊᒻᒪᓗ ᐃᓕᖕᓄᑦ ᐅᑎᕐᕕᒋᔫᕐᓗᓯ.
- **ᐊᓖᓴ**: ᑕᒪᓐᓇ ᐅᖃᐅᓯᕆᔪᓐᓇᕐᒥᔭᐃᑦ ᑲᑎᒪᔨᐅᓪᓗᑕ ᑲᑎᒪᔪᓐᓇᕈᑦᑕ. ᑐᓂᓯᔪᓐᓇᖅᑐᑎᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐋᖅᑭᒋᐊᕐᓗᒍ ᑲᑎᖕᓱᕆᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᑐᒃᓯᕋᖅᑕᖅ. ᐱᒻᒪᓇᐅᓇᔭᖅᑐᖅ ᐃᓚᐅᑎᓪᓗᒋᑦ ᐃᓚᖏᑦ ᐱᔭᑎᑦ ᖃᓄᐃᒻᒪᓪᓕ ᐊᓯᔾᔨᕈᒪᒪᒻᒪᖔᑦ ᓲᖃᐃᒻᒪ ᐃᑲᔪᕐᒪᑦ ᑲᑎᒪᔨᓂᑦ (NWMB) ᑐᑭᓯᔪᓐᓇᖅᓯᓗᑎᒃ ᑭᓯ ᐱᔾᔪᑕᐅᖕᒪᖕᒥᑦ.
- **ᑲᐃᔫ:** ᑲᑎᒫᑦ (NWMB) ᐃᓚᒋᔭᐅᔫᑦ ᖁᔭᓕᑦᑎᐊᖅᑐᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᓱᒪᒋᔭᕐᒥᓐᓂᑦ ᑖᒃᑯᓂᖓ ᐃᓱᒪᓕᐅᕆᓂᐊᕐᓗᑎᒃ. ᐊᓯᖏᓐᓂᑦᑕᐅᖅ ᐱᕕᒃᓴᖃᕐᓂᐊᖅᑐᒍᑦ ᑕᒪᓐᓇ ᐅᖃᐅᓯᕆᔭᐅᓗᒍ ᑲᑎᒪᓂᑦ. ᐅᑐᐱᕆᒥ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᒫᓐᓇ ᓴᖅᑭᑦᑎᓖᖅᐸᑦ ᑖᒃᑯᓂᖓ ᑲᑎᒪᓂᑦ. ᑲᑎᒫᑦ ᑲᑎᒪᒌᖅᑐᑦ ᐅᑐᐱᕆᒥ, ᐊᒻᒪᓗ ᐅᐸᒍᓇᖅᑐᓯ ᓇᐅᒃᑰᓯᓗᑎᒃ ᐅᕙᖕᓄᑦᑕᐅᓂᑦ ᐅᐸᒍᑎᓗᑎᒃ, ᓇᓗᔪᖕᒪ ᓇᓂᒃᑲᓯᒪᖕᒪᖔᑦ ᑲᑎᒪᓂᖅ. ᖁᔭᓕᓯᑲᔭᖅᐸᖕ ᑲᑎᒪᓂᓄᑦ ᐱᖃᑕᐅᑎᑦᑎᓇᕈᐊᖅᐊᓕ ᐃᓱᒪᓕᐅᕈᓯᒐᓐᓂᐊᕐᓗᑎᒃ. ᑕᐃᓇᐅᑎᓗᒍ, ᐃᓱᒪᓕᐅᖃᑦᑕᖅᐸᑕ, ᓂᕈᐊᕈᓐᓇᖅᑐᑦ ᓈᓚᙵᓂᖅᓯᕐᓗᑎᒃ ᑐᑭᓯᐊᕈᑎᖅᑲᐊᖅᐸᑕ. ᓴᓂᑭᓗᐊᖅ ᐊᒃᑐᖅᑕᐅᓂᖅᐸᖕᓗᑎᓗᒍ, ᑕᒫᓂ ᓈᓚᙵᓂᖅᓯᕐᑎᑦᑎᓪᓚᓗᐃᓇᓇᐊᓕᑦ, ᑭᓯᐊᓂ ᐅᖅᑲᑦᓯᐊᕈᓘᓐᓇᕐᒥᑦᑐᖅ. ᑎᑎᖅᑲᖅ ᐃᓚᐅᓂᓯ ᐅᑐᐱᕆᒥ ᑲᑎᒪᓂᒥ ᐃᑲᔫᑎᖃᕐᒪᑕᓕᓵᖅᑐᖅ ᐃᓱᒪᓕᐅᕆᓂᖏᓐᓄᑦ.

- **ᐊᓖᓴ**: ᐃᕐᖑᓈᖅᑕᐅᒃᑯᑦ ᑎᑎᕋᕐᓂᐊᖅᐳᒍᑦ ᐊᐱᖅᑯᑎᒃᓴᓂᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔾᒃᑯᓐᓄᑦ ᐊᒻᒪ ᓘᑲᓯᐅᑉ ᑲᑎᒪᔨᓄᑦ ᐅᓂᒃᑳᐅᓯᕆᓂᐊᖅᓲᓂᒋᑦ.
- ᐊᓖᓴ ᓄᓇᖖᑐᐊᒥᑦ ᓴᖅᑮᔭᖅᑎᑦᑎᓚᐅᖅᐳᑦ ᓇᓐᓄᒃᑕᐅᓵᓚᐅᖅᑐᒥᑦ ᖁᕐᕙᓯᕈᖅᓯᔪᒥᑦ. ᖁᕐᕙᓯᕈᕐᕙ ᐊᔪᓕᓚᐅᖅᐳᖅ, ᑕᐃᒪᐃᓐᓂᕐᕙᓄᑦ ᓇᓄᕐᒥᑦ ᑎᑭᕌᖅᑐᒥᒃ ᓴᓂᑭᓗᐊᕐᒧᑦ ᓴᖅᑮᔭᖅᑐᖃᓚᐅᙱᑦᑐᖅ. ᓇᓄᐃᑦ ᐃᓅᓂᑯᑦ 2008−ᒥ ᖁᕐᕙᓯᕈᖅᑕᐅᓚᐅᖅᓯᒪᓪᓗᑎᓪᓗ ᐋᓐᑎᐅᕆᐆ−ᒫᓂᑑᐸ ᑭᓪᓕᕐᕙᑕ ᖃᓂᒌᔭᓂ. ᑭᖑᓪᓕᖅᐹᒥ ᖃᐅᔨᒪᔭᐅᔪᖅ ᐃᓂᒋᔭᐅᓚᐅᖅᑐᖅ ᒪᐃ 2023−ᒥ ᑭᓯᐊᓂ ᖃᐅᔨᒪᙱᕐᕋᓂ ᖃᓄᖅ ᐊᑯᓂᐅᑎᒋᔪᖅ ᑎᑎᕋᖅᓯᒪᔪᑦ ᑕᑯᒃᓴᐅᑎᑦᑎᓇᔭᕐᒪᕝᑕ.
- ᐊᓖᓴ ᓇᒃᓯᐅᑦᔾᔪᒪᓚᐅᖅᑐᖅ ᖁᕐᕙᓯᕈᖅᓱᒃᑯᑦ ᑎᑎᖅᑐᒐᕐᓂᒃ ᑕᓯᐅᔭᕐᔪᐊᑉ ᐱᖕᒐᓐᓇᖅᐸᓯᐊᓂ ᓇᓄᕐᓂᑦ ᓘᑲᓯᒧᑦ ᑕᑯᖅᑯᑎᔭᒃᓴᖏᓐᓂᒃ ᑲᑎᒪᔨᓄᑦ.

ᐃᓚᐅᖃᑕᐅᔪᑦ ᖁᔭᓕᔭᐅᓪᓗᑎᒃ ᐅᐸᒍᑎᓂᖏᓐᓂᒃ, ᑐᓴᖅᑎᑦᑎᓂᖏᓐᓂᓪᓗ.

21:30 ᑲᑎᒪᓂᖅ ᐃᓱᓕᑉᐳᖅ

-ᐊᖏᕈᑎᖃᖅᐲ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦ ᑕᐃᒪᐃᓪᓗᐊᓴᐃᓐᓇᑐᐃᓐᓇᑑᔫᕐᒪᑕ? ᐋ'

ᓈᒻᒪᐹᑦ ᓇᓄᐃᑦ ᐆᒪᔪᖅᑕᐅᕙᑦᑐᑦ? ᐃᑭᓗᐊᖅᐹᑦ? ᐅᓄᓗᐊᖅᐹᑦ?ᐋ', ᐃᑭᓗᐊᖅᑐᑦ ᓇᓄᐃᑦ ᐱᒻᒪᕆᐅᒐᓗᐊᖅᑎᓗᒋᑦᓂᕿᒋᔭᐅᔪᒻᒪᕆᐊᓘᒐᒥᒃ, ᑭᓯᐊᓂ ᒥᑏᑦ ᐱᒻᒪᕆᐅᓂᖅᓴᐃᑦ, ᐱᑦᔪᑎᖃᖅᓱᑎᒃ ᓇᓄᐃᑦ ᓱᒃᑯᓴᐃᔪᒻᒪᕆᐊᓘᒻᒪᑕ ᐃᕙᕕᐅᕙᑦᑐᓂᑦ ᕿᑭᖅᑕᐃᑦ ᓯᕿᓂᖓ ᑕᕐᕋᖓᓂᓪᓗ ᑕᓪᓕᒪᐅᔪᖅᑐᐃᑦ ᐊᕐᖂᒍᐃᑦ ᐃᓗᐊᓂ ᑕᒐᑕᒐ.

1. ᓱᓇᓂᑦ ᓇᐅᑦᓯᑐᖅᓯᒪᕕᓯ ᓇᓄᐃᑦ ᑎᒥᖏᑦᑕ ᖃᓄᐃᓐᓂᖏᓐᓂᑦ ᐊᕐᖂᒎᖃᑦᑕᓚᐅᖅᑐᓂ? ᐋ', ᖁᐃᓂᔪᐊᓗᐃᑦ ᐊᓯᑦᔨᓯᒪᑦᔪᖏᑦᑐᑦ, ᓂᕆᕙᑦᑐᑦ ᒥᑎᕐᓂᑦ, ᓇᐅᔭᓂᑦ, ᑕᑲᑕᑭᐊᓂᑦ, ᑲᑦᔮᑲᓪᓚᓂᑦ ᐊᒻᒪᓗ ᐊᓯᖏᓐᓂᑦ ᑎᒻᒥᐊᓂᑦ ᐃᕙᕙᑦᑐᓂᑦ.
2. ᓱᓇᑐᐃᓐᓇᓂᑦ ᐱᒻᒪᕆᐅᔪᓂᑦ ᐅᖃᐅᓯᖃᕈᒪᕕᑦ ᓇᐅᑦᓯᑐᖅᓯᒪᔭᑦᓯᓂᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓴᐅᖅᑯᔭᖅᓯᓂᑦ ᐅᕙᑦᑎᓄᑦ? ᐊᐅᑲ, ᑭᓯᐊᓂ ᐊᒥᓲᓗᐊᖅᑐᐃᑦ ᓇᓄᐃᑦ.
3. ᓇᓪᓕᐊᓂ ᓇᒻᒪᓴᑉᐱᑦ/ᓈᒻᒪᓴᖕᖏᓚᑎᑦ ᖃᐅᔨᓯᒪᔪᖅᑎᓂᑦ? ᑕᐃᒪᐃᓪᓗᐊᓴᐃᓐᓇᑑᔫᖅᑐᑦ ᐅᓄᕐᓂᖏᑦ ᑭᓯᐊᓂ, ᐅᓄᕐᓂᓴᒻᒪᕆᐅᓕᖅᑐᑦ ᓇᓄᐃᑦ.

ᐅᑯᓄᖓ ᑐᓂᔭᐅᔪᖅ

ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ

ᐆᒧᖓ

ᑐᓴᖅᑕᐅᔪᒃᓴᑦ: X **ᐃᓱᒪᓕᐅᕈᑕᐅᔪᑦ:**

ᐱᔾᔪᑕᐅᔪᖅ: ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᑕ ᐅᓄᕐᓂᖏᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᖅ

ᐱᓕᕆᐊᖑᓯᒪᔪᑦ:

- ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᑦ ᐊᒥᓲᓛᖑᕗᑦ ᐅᐊᖕᓇᖓᓂ ᐅᑭᐅᖅᑕᖅᑐᒥᑦ, ᐃᓄᖃᖅᖢᑎᒃ ᐊᕐᕌᒍᑕᒫᑦ 300,000 ᑭᓚᒦᑕᓂᑦ^{2}.
- ᑎᑎᕋᖅᑕᐅᓯᒪᓚᐅᙱᑎᓪᓗᒋᑦ 495,665−ᓂᒃ ᑐᒃᑐᓂᒃ 1994−ᒥ, ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᑦ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᖅᓯᒪᖅᑰᔨᓚᐅᖅᐳᑦ 1950−ᖏᓐᓂ. ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓂᒃ ᑎᑎᕋᖅᓯᒪᕗᑦ ᐅᔾᔨᕐᓇᖅᑐᒥᒃ ᐊᓯᔾᔨᖅᓯᒪᔪᓂᒃ 1970 ᐃᓱᖅᐸᓯᐊᓂ ᒫᓐᓇᒧᑦ ᑎᑭᖢᒍ.
- 11,000−ᐸᓗᖕᓂᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᑉᐸᖕᓂᖏᓐᓂᒃ ᐊᕐᕌᒍᑕᒫᑦ ᓄᓇᕗᒻᒥ, ᓄᓇᑦᑎᐊᒥ, ᒪᓂᑑᐸᒥ ᓴᔅᑳᑦᓯᐗᒥᓗ. ᑭᕙᓪᓕᕐᒥ ᐃᓄᐃᑦ ᑐᒃᑐᓐᓇᓱᒋᔭᐅᕗᑦ 8,000 ᐅᖓᑖᓂ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓂᒃ ᐊᕐᕌᒍᑕᒫᑦ. ᑖᒃᑯᐊ ᐱᒻᒪᕆᐅᒋᔭᐅᔪᑦ ᑕᑯᒃᓴᐅᑎᑦᑎᔪᑦ ᓇᓚᐅᑦᑖᖅᓯᒪᙱᓗᐊᕐᓂᕐᒥᒃ ᓯᓚ ᓇᓗᓇᕈᔪᖕᒪᑦ ᖃᔅᓯᐅᓪᓚᑦᑖᕐᓂᖏᓐᓂᒃ ᑐᒃᑐᑕᐅᕙᒃᑐᑦ.
- ᓄᓇᕗᒻᒥ ᐊᕕᒃᑐᖅᓯᒪᓂᐅᔪᓂ ᑕᐅᖅᓲᖃᑦᑕᕐᓂᖅ ᑐᒃᑐᓂᑦ ᖃᕋᓴᐅᔭᒃᑯᑦ ᐊᖑᓇᓱᒍᐊᖅᑎᑦᑎᓕᕐᒪᑦ ᐱᓗᐊᖅᑕᐃᓕᓂᕐᒧᓪᓗ ᐃᓱᒫᓘᑕᐅᓕᖅᖢᓂ ᑭᕙᓪᓕᖅ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ. ᑭᕙᓪᓕᕐᒥ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐃᓱᒫᓘᑎᖃᖅᐳᑦ ᓄᓇᕗᒻᒥ ᐊᕕᒃᑐᖅᓯᒪᓂᐅᔪᓂ ᓂᕿᓂᒃ ᓂᐅᕐᕈᑎᖃᖅᐸᒃᑐᑦ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᐊᔾᔪᑕᐅᖏᓐᓇᕐᓂᐊᕐᓂᖓᓂᒃ. ᑕᒪᓐᓇ ᒫᓐᓇ ᖃᐅᔨᓴᖅᑕᐅᕙᓪᓕᐊᙱᓚᖅ ᐅᖃᐅᓯᐅᙱᖅᖢᓂᓗ.
- ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᐅᑎᓕᕆᓂᖅ ᐱᒋᐊᖅᑎᑕᐅᓚᐅᖅᓯᒪᔪᖅ 1993-ᒥ ᐊᒻᒪᓗ ᐅᓪᓗᒥ ᑲᔪᓯᓪᓗᓂ. ᑖᓐᓇ ᐱᓕᕆᐊᖑᔪᖅ ᐃᑲᔪᖅᑐᐃᓯᒪᕗᖅ ᓴᖅᑭᑎᑦᑎᓂᕐᒥᒃ ᐃᓗᓕᖅᑯᑦᑎᐊᖅᑐᒥᒃ ᐃᓂᒥᒃ ᐱᓕᕆᐊᖑᔪᓄᓪᓗ ᑎᑎᕋᖅᓯᒪᔪᖁᑎᒋᔭᐅᔪᓂᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑑᖃᑎᒌᓄᑦ. ᑖᓐᓇ ᓄᐊᑦᑎᕝ ᐱᕕᑐᐊᑦᑎᐊᖑᔪᖅ ᐅᐊᓕᓂᕐᒥᑦ ᖃᐅᔨᓴᖅᑎᒻᒪᕆᓐᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᖃᐅᔨᓴᖅᑐᑎᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᑦᑐᓂᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᕐᕌᒍᑕᒫᑦ.
- ᖃᖓᑦᑎᐊᓕᓴᑦ ᓇᒧᙵᖅᐸᓪᓕᐊᓂᖏᑦ (ᐅᑭᐅᑦ ᑕᓪᓕᒪᑦ ᐃᓗᐊᓂ) ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓂᒃ ᓄᕐᕌᖏᑦ ᓇᑐᓇᐃᖅᓯᓯᒪᕗᑦ ᓄᕐᓇᐅᕐᕕᖏᓐᓂᒃ ᐊᓯᔾᔨᖅᐸᓐᓂᖏᓐᓂᒃ, ᑕᑉᔪᐊᖅᑐᑦ ᐊᖅᑯᑎᖏᓐᓂᒃ, ᐅᐱᕐᖔᒃᑯᑦ, ᐊᒻᒪ ᐊᐅᔭᒃᑯᑦ ᓇᔪᖅᐸᒃᑕᖏᓐᓂᒃ ᓄᓇᒥᓐᒫᖅᑐᓂᒃ ᐱᕙᓪᓕᐊᑎᑕᐅᔪᑦ ᐊᐅᓚᔭᓐᓇᐅᑎᖏᓐᓂᒃ. ᐊᒻᒪ ᓱᓕ, ᐅᑭᐅᒃᑯᑦ ᖃᒪᓂᕐᔪᐊᖅ, ᕿᙵᐅᑦ ᐊᒻᒪ ᑎᔅᓯᐅᑉ ᑐᒃᑐᖏᑦ ᖃᓪᓕᖅᓯᒪᕐᔪᐊᖅᐳᑦ ᐅᑭᐅᑦ ᖁᓕᑦ ᐃᓗᐊᓂ.

ᒫᓐᓇ ᖃᓄᐃᓕᖓᓂᖓ

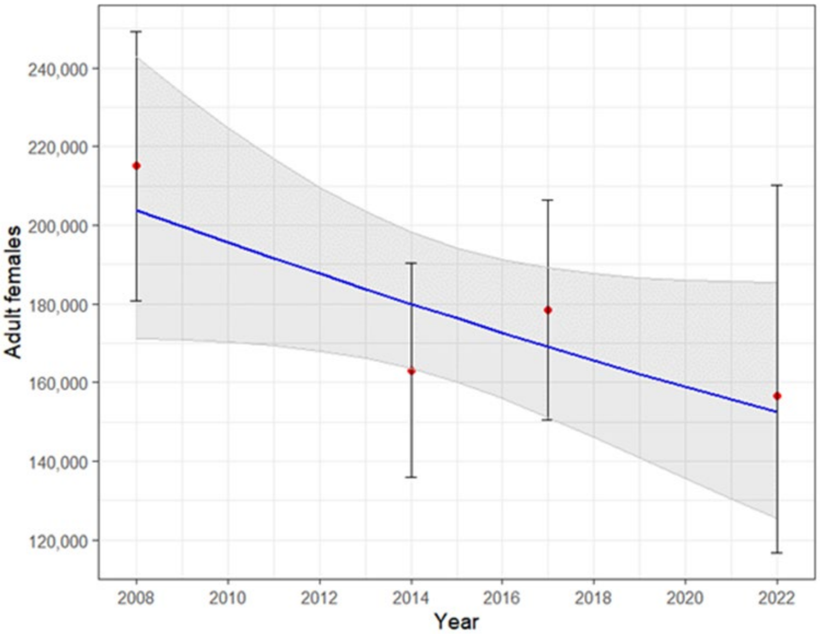
- ᔫᓂ 2022−ᒥ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᖏᑦ ᐱᐊᓂᒃᓯᓚᐅᖅᐳᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓂᒃ ᒥᒃᓴᐅᑦᑎᓇᓱᒃᖢᑎᒃ ᐃᓐᓇᐃᑦ ᐊᕐᓇᓪᓗᐃᑦ ᐊᒻᒪ ᑐᒃᑐᓕᒫᑦ.

- ᖃᐅᔨᓴᕈᑎᕕᓃᑦ 156,540−ᖑᓇᓱᒋᔭᐅᓚᐅᖅᐳᑦ ᐊᕐᓇᓪᓗᐃᑦ, 213,079−ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ ᑐᒃᑐᐃᑦ ᓄᕐᕆᐅᕐᕕᖏᓐᓂᑦ ᐊᒻᒪᓗ ᑲᑎᐅᖅᑲᔪᑦ ᓇᓚᐅᑦᑖᖅᖢᑎᒃ 252,892−ᓂᑦ ᐃᓐᓇᖕᓂᑦ ᑐᒃᑐᓂᑦ.

- ᑖᒃᑯᐊ ᖃᓄᐃᓕᖓᓂᖏᑦ ᓇᓗᓇᐃᖅᓯᕗᑦ ᑐᒃᑐᐃᑦ ᓱᕐᕌᓗᐊᔾᔫᕋᓱᒋᔭᐅᙱᖦᖢᑎᒃ. ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ 2022−ᒥ ᐊᒻᒪ 2008−ᒥ ᓇᓗᓇᐃᖅᓯᕗᑦ ᓈᓴᖅᑕᐅᓯᒪᔪᑦ ᐅᔾᔨᕐᓇᖅᑐᒥᒃ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕐᓂᖏᓐᓂᒃ. ᑭᓯᐊᓂ, ᖃᓄᐃᓕᕚᓪᓕᐊᓂᕆᔭᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᑖᒃᑯᓇᓂ ᐊᑐᖅᑎᓪᓗᒍ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᐳᑦ ᐅᔾᔨᕐᓇᓗᐊᙱᑦᑐᓂᒃ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᐊᓂᖏᓐᓂᒃ (ᐊᔾᔨᙳᐊᖅ 1).

- ᐱᑕᖃᑦᑎᐊᙱᓐᓂᖓ ᑭᒡᓕᓯᓂᐊᖅᑕᐅᓯᒪᔪᓄᑦ ᐅᔾᔨᕐᓇᕐᓂᕆᔭᐅᔪᑦ ᐃᓛᒃᑰᓕᖅᐸᖕᓂᖏᓄᑦ−ᑐᙵᕕᓖᑦ ᖃᓄᐃᓕᕚᓪᓕᐊᓂᕆᔭᐅᔪᑦ ᖃᔅᓯᐅᓇᓱᒋᔭᐅᔪᑦ (ᑕᒪᓐᓇ ᐃᓚᖃᖅᖢᓂ 2008, 2014, 2017−ᒥ, 2022−ᓂᓪᓗ ᖃᔅᓯᑐᕐᓂᐊᕋᓱᒋᔭᐅᔪᓂᒃ) ᑕᐃᒪᐃᔾᔪᑎᖃᑐᐃᓐᓇᓇᐊᖃᖅᐳᖅ ᐊᑕᐅᔩᓐᓇᓕᓯᒪᓂᖏᓐᓄᑦ 2014-2017−ᒥ, ᑭᓯᐊᓂᓗᑦᑕᐅᖅ, ᐅᔾᔨᕆᔭᖃᓪᓗᐊᖅᐳᖅ ᖃᔅᓯᐅᓇᓱᒋᓈᓂᕐᓂᒃ (ᐊᖏᔪᒻᒪᕆᖕᓂᒃ ᓇᓗᙱᑦᑎᐊᕐᓂᕐᒧᑦ ᖃᖓᒃᑰᕋᔭᕐᓂᖏᓂᒃ) ᐊᔪᕐᓇᖅᑎᑦᑎᑐᐃᓐᓇᓇᐊᖃᖅᐳᖅ ᑐᔾᔭᖅᓯᔭᕆᐊᒃᓴᖅ ᐅᓄᙱᓂᖅᓴᓂᒃ ᐊᓯᔾᔨᕐᓂᕆᔭᐅᔪᓂᒃ ᖃᓄᖅ ᐅᓄᖅᑎᒋᓂᖏᓄᑦ ᖃᐅᔨᓴᕐᓂᓴᐅᔪᒥᑦ.

- ᔫᓂ 2022−ᒥ ᓇᐃᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ ᐃᓗᐃᑦᑐᒥᒃ 35,352 ᐃᓐᓇᐃᑦ ᐊᒻᒪ ᓄᕐᕋᑯᓗᐃᑦ ᐅᔾᔨᕆᔭᐅᓚᐅᖅᑐᑦ ᓇᐃᓴᖅᑕᐅᔪᑦ ᔫᓂ 2017−ᒥ ᐅᓄᙱᓂᖅᓴᐅᕌᖅᔪᓕᕐᓂᖏᓂᒃ. ᑭᒡᓕᓯᓂᐊᕈᑎᓄᑦ ᐱᒻᒪᕆᐅᙱᒃᑲᓗᐊᖅᖢᓂ, ᑖᓐᓇ ᒥᒃᓴᒥᐊᖅᑕᐅᔪᖅ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᖅ ᓇᓗᓇᐃᖅᓯᕗᖅ ᐱᒻᒪᕆᐅᓂᖓᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᙱᓐᓇᓂᐊᖅᑲᓂᖅᐸᑦ ᑐᒃᑐᐃᑦ.



ᐊᔾᔨᙳᐊᖅ 1. ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᑦ ᐅᓄᕐᓂᖏᑦ ᐃᓱᒃᐸᓪᓕᐊᔪᓄᑦ ᐃᓚᓕᐅᔾᔭᐅᓪᓗᑎᒃ ᔫᓂ 2022−ᒥ ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ. ᖁᓯᙱᓐᓂᖅ ᑭᒡᓕᒋᑎᑕᐅᔪᑦ ᑲᑎᙵᔪᓐᓃᖅᑐᓄᑦ ᓇᓚᐅᑦᑖᕈᑕᐅᔪᓄᑦ ᑐᓂᔭᐅᕙᒃᐳᑦ ᑐᖑᔪᖅᑐᓂᒃ ᑐᑭᓕᐊᖅᑐᓂᒃ. ᐃᓛᒃᑯᑦ ᓇᓚᐅᑦᑖᓚᑦ ᓴᖅᑮᔭᖅᑐᑦ ᐊᐅᐸᖅᑐᑦ 95 ᐳᓴᓐᑎᒥᑦ ᖁᓯᙱᓐᓂᕐᒧᑦ ᐃᓱᓕᕝᕕᖏᑦ.

ᑐᓴᕆᐊᕐᕕᖃᕐᓂᖅ

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖏᑦ ᓄᓇᓕᖕᓄᐊᓚᐅᖅᐳᑦ ᑭᕙᓪᓕᕐᒥ ᔭᓄᐊᕆᒥ ᕕᕗᐊᕆᒥᓗ 2024−ᒥ ᑲᑎᖃᑎᖃᓇᐊᖅᑐᖅᓱᑎᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓂᒃ ᓄᑖᙳᓇᐊᖅᓯᒪᔪᓂᓪᓗ ᑐᓴᖅᑎᑦᑎᔭᖅᑐᖅᓱᑎᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓂᒃ ᖃᐅᔨᓴᖅᓯᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᐅᓕᖅᑐᓂᒃ, ᐊᓯᖏᓐᓂᓪᓗ ᐊᕕᒃᓯᒪᓂᕐᓂ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᐅᓕᖅᑐᓂᒃ ᐃᓱᒪᒋᔭᒃᓴᓂᓪᓗ. ᐅᓂᒃᑳᓇᐊᖅᑐᖅᓯᒪᔪᑦ ᓇᐃᓈᖅᓯᒪᓪᓗᑎᒃ ᖃᐅᔨᓴᕈᑎᖏᑦ ᖃᖓᑦᑎᐊᖅ ᓈᓴᖅᑕᐅᓚᐅᖅᑐᓂᑦ ᐊᓯᖏᓪᓗ ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᑭᕙᓪᓕᕐᒥ ᑐᒃᑐᐃᑦ ᐱᔾᔪᑎᒋᓗᒋᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᑐᓂᔭᐅᓚᐅᖅᐳᑦ.

ᐊᑐᓕᖁᔭᐅᓯᒪᔪᖅ

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑐᒃᓯᕋᙱᑦᑐᑦ ᖃᓄᑐᐃᓐᓇᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᖃᓄᐃᓕᐅᕈᑎᓂᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓄᑦ ᒫᓐᓇᐅᔪᖅ ᑭᓯᐊᓂ ᐱᖁᔨᔪᑦ ᖃᐅᔨᓴᐃᓐᓇᕐᓗᑎᒃ ᑖᒃᑯᓂᖓ ᑐᒃᑐᓂᒃ.

Estimating Abundance of the Qamanirjuaq Mainland Migratory Barren-Ground Caribou Subpopulation - June 2022

FILE REPORT

Government of Nunavut
Department of Environment
Technical Report Series – No: 01-2023

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***Disclaimer**:*
The conclusions and recommendations discussed within this report represent those of the authors and do not necessarily reflect the position of the Government of Nunavut.

ᖃᐅᔨᓴᕐᓂᖅ ᓇᐃᓈᖅᓯᒪᔪᖅ

ᒫᓐᓇᓕᓴᑦ ᓄᓇᐃᓐᓇᕐᒥ ᑐᒃᑐᐃᑦ ᐃᕐᓂᐅᕐᕕᖏᓐᓂᒃ ᖃᐅᔨᓴᓲᖑᕗᑦ ᐊᕐᓇᓪᓗᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᒃ (ᐊᒻᒪ ᓈᒻᒪᒃᑐᓂᒃ ᐃᕐᓂᐅᖅᑐᓂᒃ ᐊᕐᓇᓪᓗᖕᓂᒃ) ᑲᑎᖦᖢᒋᑦ ᐅᑭᐊᒃᓵᒃᑯᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ, ᒥᒃᓴᐅᑦᑎᓇᓱᒃᖢᑎᒃ ᖃᐅᔨᓇᓱᓪᓗᒋᓪᓗ ᑐᒃᑐᐃᑦ ᐅᓄᕐᓂᖏᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᓪᓗ. ᔫᓂ 2008–ᒥᑦ ᔫᓂ 2017–ᒧᑦ, ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᓄᓇᐃᓐᓇᕐᒥ ᑕᒡᔪᐊᖅᐸᒃᑐᑦ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖏᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᓪᓗ ᒪᓕᓚᐅᖅᐳᑦ ᐊᔾᔨᐸᓗᖏᓐᓂᒃ ᐱᓕᕆᔾᔪᓯᕐᓂᒃ. ᔫᓂ 2022–ᒥ, ᓇᐃᓴᐃᔭᖅᑐᕋᓯᓚᐅᖅᓯᒪᕗᒍᑦ ᐊᕐᓇᓪᓗᐃᑦ ᐅᓄᕐᓂᕆᓪᓗᐊᖅᑕᖏᓐᓂᒃ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᕐᑯᑎᖏᑦ ᑖᒃᑯᐊᓗ ᓇᐃᓴᖅᑕᐅᓂᕆᔭᖏᑦ ᐊᑐᖅᑕᐅᓕᓚᐅᖅᓯᒪᓪᓗᑎᒃ ᐃᓗᐃᑦᑐᐃᑦ ᑕᒡᔪᐊᔪᑦ ᖃᔅᓯᐸᓘᓪᓗᐊᕐᒪᖔᑕ ᐅᑭᐊᒃᓵᒃᑯᑦ ᓇᐃᓴᖅᑕᐅᓕᖅᑎᓪᓗᒋᑦ ᖃᐅᔨᓴᕈᑕᐅᓚᐅᖅᑐᑦ. ᔫᓂ 2008-ᒥ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᒥᒃᓴᐅᑦᑎᓚᐅᖅᐳᑦ 215,049 (95% CI=180,770-249,328; CV=8.1%) ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᑐᒃᑐᐃᑦ ᖃᒪᓂᕐᔪᐊᖅᒥ ᐊᕐᕌᒍᑕᒫᑦ ᓄᕐᕋᐅᕐᕕᒋᕙᒃᑕᖓᓂ, ᑲᑎᖦᖢᒋᑦ ᑲᑎᖦᖢᒋᑦ ᖃᔅᓯᐅᓇᓱᒋᔭᐅᓂᖏᑦ 344,078 (95% CI=287,208-400,948; CV=8.1%) ᐃᓐᓇᐃᑦ ᑐᒃᑐᐃᑦ. ᔫᓂ 2014-ᒥ, ᖃᐅᔨᓴᕐᓂᖅ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᖅ 163,066 (95% CI=136,317-189,815; CV=8.2%) ᐃᓐᓇᐃᑦ ᐊᕐᓇᓪᓗᐃᑦ ᑐᒃᑐᐃᑦ ᑲᑎᖦᖢᒋᑦ ᖃᔅᓯᐅᓇᓱᒋᔭᐅᓂᖏᑦ 264,718 (95% CI=220,634-308.802; CV=8.3%) ᐃᓐᓇᐃᑦ ᑐᒃᑐᐃᑦ. 2014-ᒥ ᖃᐅᔨᓴᕈᑎᕙᓃᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᐳᑦ ᐅᔾᔨᕐᓇᖅᑐᒥᒃ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕐᓂᖏᓐᓂᒃ (DF=71.3; T=-2.23; P=0.029) ᐊᑯᓐᓂᖏᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ 23% ᐊᑦᑎᓪᓕᒋᐊᖅᑐᑦ 6–ᓄ ᐊᕐᕌᒍᓄᑦ. ᑕᑯᔭᐅᔪᑦ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᖅᑎᓪᓗᒋᑦ, ᑐᒃᑐᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᒃᑲᓐᓂᓚᐅᖅᐳᑦ ᔫᓂ 2017–ᒥ ᓇᓚᐅᑦᑖᕈᑕᐅᓚᐅᖅᓯᒪᓪᓗᓂ 178,423–ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ (95% CI=150,468-206,377; CV=7.6%) ᐃᓐᓇᐃᑦ ᐊᕐᓇᓪᓗᐃᑦ ᑐᒃᑐᐃᑦ, ᑲᑎᖦᖢᒋᑦ ᖃᔅᓯᐅᓇᓱᒋᔭᐅᓂᖏᑦ 288,244 (95% CI=242,121-334,367; CV=7.8%) ᐃᓐᓇᐃᑦ ᑐᒃᑐᐃᑦ ᓇᓗᓇᐃᖅᐳᖅ ᐅᔾᔨᕐᓇᓗᐊᙱᑦᑐᒥᒃ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᐊᓂᖏᓐᓂᒃ ᐊᕐᕌᒍᑕᒫᑦ 0.98-ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ (CI=0.94-1.01). ᔫᓂ 2022 (ᑖᓐᓇ ᐅᓂᒃᑳᖅ) ᐊᒥᓲᓂᖏᓐᓂᑦ ᓇᐃᓴᐃᓂᖅ ᓇᓚᐅᑦᑖᕈᑕᐅᓚᐅᖅᐳᖅ 156,540–ᓂᑦ ᐃᓐᓇᑦ ᐊᕐᓇᓪᓗᐃᑦ

(95% CI=116,635-210,099; CV=13.8%) ᑲᑎᖅᓱᒋᑦ ᑐᒃᑐᐃᑦ ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ 252,892 (95% CI=188,050-340,092: CV=13.9%) ᐃᓐᓇᐃᑦ ᑐᒃᑐᐃᑦ. ᑲᑎᖅᓱᒋᑦ ᐅᓄᕐᓂᖏᑦ ᑐᒃᑐᐃᑦ (ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᐸᖕᓃᑦ ᑎᕿᑐᕋᐃᓪᓗ) ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ ᓄᕐᕆᐅᕐᕕᖕᓂ 262,272 (95% CI=227,910-296,634; CV=6.4%) ᔫᓂ 2014-ᒥ ᐊᒻᒪ 252,060-ᒥ (95% CI=220,721-283,398; CV=6.1%) ᔫᓂ 2017–ᒥ, ᒥᒃᓴᐅᖅᓱᒋᑦ 213,079 (95% CI=166,781-272,229; CV=11.5%) ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ ᓄᕐᕆᐅᕐᕕᖏᑦ ᔫᓂ 2022-ᒥ. ᐊᑕᖏᖅᓱᒍ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ ᓇᓗᓇᐃᖅᓯᕗᑦ ᑐᒃᑐᐃᑦ ᓱᕐᕌᓗᐊᖅᑑᔨᙱᖅᓱᑎᒃ ᐅᓄᕐᓂᖏᑎᒍᑦ. ᒥᒃᓴᐅᖅᓱᒋᑦ 2022-ᒥ ᓴᓂᐊᓂᓕ 2008-ᒥ ᒥᒃᓴᐅᑦᑎᓯᒪᔪᑦ, ᓇᓗᓇᐃᖅᓯᕗᑦ ᐅᔾᔨᕐᓇᖅᑐᒥᒃ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕐᓂᖏᓐᓂᒃ. ᑭᓯᐊᓂ, ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐅᓄᕐᓂᕆᓚᐅᖅᑕᖏᑦ 2008 ᐊᒻᒪ 2022 ᐊᑯᓐᓂᖓᓂ ᐊᓯᔾᔨᓗᐊᙱᑦᑐᖅ. ᔫᓂ 2022–ᒥ ᓇᐃᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ ᐃᓗᐃᑦᑐᒥᒃ 35,352 ᐃᓐᓇᐃᑦ ᐊᒻᒪ ᓄᕐᕋᑯᓗᐃᑦ ᐅᔾᔨᕆᔭᐅᓚᐅᖅᑐᑦ ᓇᐃᓴᖅᑕᐅᔪᑦ ᔫᓂ 2017–ᒥ ᐅᓄᙱᓐᓂᖅᓴᐅᕚᖅᔪᓕᕐᓂᖏᓐᓂᒃ. ᑭᓪᓕᓯᓂᐊᕈᑎᓄᑦ ᐱᒻᒪᕆᐅᙱᒃᑲᓗᐊᖅᓱᓂ, ᑖᓐᓇ ᒥᒃᓵᕆᐊᖅᑕᐅᔪᖅ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᖅ ᓇᓗᓇᐃᖅᓯᕗᖅ ᐱᒻᒪᕆᐅᓂᖓᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᖏᓐᓇᕆᐊᖃᕐᓂᖏᑦ ᑐᒃᑐᐃᑦ. ᓇᓗᓇᐃᖅᑕᐅᖅᑲᐅᕗᖅ, ᖃᑯᑎᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᒃᑲᓐᓂᖃᑦᑕᕆᐊᖃᖅᐳᑦ ᑐᒃᑐᐃᑦ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᔭᐅᖁᓪᓗᒋᑦ ᐊᒻᒪ ᑲᒪᒋᔭᐅᓗᑎᒃ ᐊᐅᓚᑦᑎᔨᐅᖃᑕᐅᔪᓂᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᒥᑦ ᓄᓇᕗᒻᒥ ᐊᖏᕈᑎᒥᑦ. ᐊᔾᔨᖓᓂᒃ ᐊᒃᓱᕈᕐᓇᖃᓯᐅᑎᕗᖅᑕᐅᖅ [illegible] ᓄᓇ ᐊᖏᕐᕋᓂᖅᑖᖅ ᑕᒪᓂᓯᐅᕐᓗᑎᒃ ᓇᓄᕈᓇᑦᑎᐊᕐᓗᑎᒃ ᐅᓄᖅᓯᕙᓪᓕᐊᔪᓐᓇᖃᔭᕐᒪᑕ ᓯᓚ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖅᑯᑎᖏᑦ. ᑕᒪᓐᓇ ᐱᒻᒪᕆᐊᓗᒃᔪᖅ ᓄᓇᖅᐸᓯᒃᑐᒥ ᑕᒃᔪᐊᖅᐸᒃᑐᓄᑦ ᖃᒪᓂᕐᔪᐊᖅ ᑐᒃᑐᖅᑯᑎᖏᓐᓄᑦ ᓱᕐᓗ ᐃᓕᖅᑯᓯᑐᐃᓐᓇᐅᔪᓐᓇᖅᓱᓂ ᐅᕙᓗᓃᑦ ᖃᓄᐃᑦᑐᖅᑯᖅᑐᕕᓂᐅᓂᕐᒧᑦ ᐅᓄᕈᓐᓃᒃᑲᓪᓚᓚᓗᐊᖅᓯᒪᓂᖅᐸᑕ.

ABSTRACT

Modern mainland migratory caribou calving ground photo surveys generally rely on the assessment of females (and where appropriate, breeding females) combined with fall composition surveys, to estimate and track herd abundance and trend. From June 2008 through June 2017, assessments of the mainland migratory Qamanirjuaq herd abundance and trend, have followed similar methods. In June 2022, we set out to estimate the abundance and trend of females in the Qamanirjuaq herd of barren ground caribou which were then used as one component to generate a whole herd estimate using fall composition studies. In June 2008, the Government of Nunavut estimated 215,049 (95% CI=180,770-249,328; CV=8.1%) adult female caribou on the Qamanirjuaq annual core calving ground, yielding a whole herd estimate of 344,078 (95% CI=287,208-400,948; CV=8.1%) adult caribou. In June 2014, the survey estimated 163,066 (95% CI=136,317-189,815; CV=8.2%) adult female caribou with a whole herd estimate of 264,718 (95% CI=220,634-308.802; CV=8.3%) adult caribou. The 2014 results confirmed a significant decline (DF=71.3; T=-2.23; P=0.029) between survey periods, indicating a 23% decline over the 6-year period. Following up on these observed declines, the herd was again surveyed in June 2017, which generated an estimate of 178,423 (95% CI=150,468-206,377; CV=7.6%) adult female caribou, and a whole herd estimate of 288,244 (95% CI=242,121-334,367; CV=7.8%) adult caribou, suggesting a non-significant decreasing trend with a yearly λ estimate of 0.98 (CI=0.94-1.01). The June 2022 (this report) abundance survey generated an estimated 156,540 adult female caribou (95% CI=116,635-210,099; CV=13.8%), which yielded a whole herd estimate of 252,892 (95% CI=188,050-340,092: CV=13.9%) adult caribou. The total number of caribou (including bulls and yearlings) estimated on the calving ground was 262,272 (95% CI=227,910-296,634; CV=6.4%) in June 2014 and 252,060 (95% CI=220,721-283,398; CV=6.1%) in June 2017, compared to 213,079 (95% CI=166,781-272,229; CV=11.5%) estimated on the calving ground in June 2022. The overall survey results suggest that the herd appears to be relatively stable. Comparison of the 2022 to the 2008 estimate directly, indicates a significant decline. However, an analysis of regression between 2008 and 2022 is not significant. The mean of the whole herd estimate of the June 2022 survey is 35,352 adult and yearling caribou below that observed in June 2017. Though not statistically significant, this reduction in the mean estimate highlights the importance of continued monitoring of the herd. As indicated, periodic re-assessments of herd abundance are necessary to ensure that any declines can be detected and addressed by co-management partners as outlined within the Nunavut Agreement. Of equal importance is the protection of critical range to ensure healthy seasonal range remains available and accessible to Qamanirjuaq caribou. This is of primary importance to mainland migratory barren-ground caribou such as the Qamanirjuaq herd when recovering from cyclical or major declines in abundance.

Key Words: Calving Ground, Photographic Survey, Mainland Migratory Caribou, Kivalliq Region, Barren-Ground Caribou, Qamanirjuaq Herd, Nunavut, *Rangifer tarandus groenlandicus*, Population Survey.

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1.0 INTRODUCTION

The Qamanirjuaq Caribou Herd is the largest herd in the western arctic, occupying an estimated annual range of 300,000 km^2. Caribou are an extremely important form of food security, and are socially and culturally important to the well being of the indigenous peoples of Nunavut, Northwest Territories, Manitoba, and Saskatchewan. The most recent estimates of caribou use (harvest) suggest Kivalliq Inuit utilize over 8,000 Qamanirjuaq caribou per year, followed by Manitoba Dene utilizing an estimated 2,000 caribou per year, and Saskatchewan and NWT range-based communities harvesting an unverified number of caribou per year. Both Saskatchewan and NWT aboriginal harvesters are thought to utilize approximately 500 to 1,000 animals annually (InterGroup, 2013). Within Nunavut, NWT, Manitoba, and Saskatchewan, indigenous harvesters are not required to report their barren-ground caribou harvest, resulting in insufficient data to quantify the harvest across the entire annual range of the Qamanirjuaq herd. Given these limitations, we relied on the 2004 Nunavut Wildlife Harvest study, and anecdotal reports gathered by the Beverly and Qamanirjuaq Caribou Management Board (BQCMB), to approximate the harvest (Priest and Usher, 2004). In total, an estimated 11,000 Qamanirjuaq caribou are harvested annually, which would produce an annual dollar equivalent of over fifteen (15) million dollars in food value to indigenous harvesters. These values likely represent an underestimate as there remains uncertainty around the actual number of caribou that are harvested. Within Nunavut, internet sales of caribou meat to other Nunavut communities are occurring but is not currently tracked or reported. Regardless of the accuracy of the estimated subsistence harvest, any decline in productivity, or increase in mortality herd wide, would have a significant impact on thousands of indigenous peoples across the Qamanirjuaq range.

A satellite telemetry program initiated in 1993 has supported the creation of a comprehensive location and activity database for the Qamanirjuaq herd. This database has been providing biologists, Hunter and Trapper Organizations (HTOs), Regional Wildlife Organization (RWOs), and inter-jurisdictional and jurisdictional management boards with the only source of western scientific based information examining the Qamanirjuaq herds use of their annual range.

Recent movements (within the last 5 years) of Qamanirjuaq caribou cows have indicated some shifts in calving-grounds, migratory corridors, spring staging areas, and summer range in the vicinity of resource development infrastructure. Additionally, Qamanirjuaq winter range over the last ten years has shown considerable overlap with the Bathurst and Beverly populations of barren ground caribou (Campbell et al. 2012-A; Campbell et al. 2014).

Trend analysis of the Qamanirjuaq herd across the June 1994, 2008, 2014, and 2017 calving-ground photographic surveys indicates a declining trend (Boulanger et al. 2018; Campbell et al. 2015-A; Campbell et al., 2010). Prior to the recorded peak in Qamanirjuaq herd abundance in the mid-1990s, the Qamanirjuaq herd showed signs of decline in the early 1950's (Parker, 1972). These early findings led to an increase in scientific studies attempting to understand the underlying mechanisms responsible for the observed declines (Heard, 1985; Parker, 1972). Research interest and efforts reached a peak between the mid 1970's and late 1980's until the results of a June 1982 survey showed that the downward trend had reversed and herd abundance was increasing (Gates, 1985). This unexpected increase in abundance was not surprising to local hunters as the local knowledge of the time disagreed strongly with the late 1970s survey estimates.

Abundance surveys of the Qamanirjuaq Herd have documented considerable change from late 1970s to present. Abundance survey findings suggest the herd dropped from an estimated 43,800 caribou in 1976 to a historical low of 39,000 adult and yearling caribou in June 1980 (Heard, 1981; Gates, 1983). These earlier surveys, however, were visual only, carrying concerns that at high densities, characteristic of the Qamanirjuaq concentrated calving ground, single observer visual estimates could be imprecise. To correct this problem, Biologists developed the June barren-ground caribou calving photographic survey method first deployed on the Qamanirjuaq herd in June 1983. The new photographic method indicated increases in abundance over the 1970's single observer visual methods and has been used since. By 1988 the herd was estimated to have increased to 221,000 (SE = 72,000), and by 1994, to 495,665 (SE = 105,426), the highest recorded abundance for the herd. By June 2008 the Qamanirjuaq subpopulation was estimated to be 344,078 (SE = 44,861) adults and yearlings (Russell, 1990; Williams, 1995; Campbell et al., 2010) which represented a significant decline from the June 1994 estimate.

From June 1983 to present, estimates of Qamanirjuaq herd size have been based on a combination of aerial photography and visual assessments of the calving ground, where the numbers of breeding cows are counted and herd abundance extrapolated using fall composition counts (**Table 1**). Up until 1994 the herd appeared to have been growing. Evidence of a decline was first detected In June 2008, at which time the Government of Nunavut estimated 344,078 (95% CI=287,208-400,948; CV=8.1%) adult caribou. A second survey flown in June 2014 estimated 264,718 (95% CI=220,634-308,802; CV=8.3%) adult caribou. The reduction in abundance between June 2008 and June 2014 was significant (DF=71.3; T=-2.23; P=0.029) suggesting a 23% decline over the 6 years between estimates. A survey flown in June 2017, estimated 288,244 (95% CI=242,121-334,367; CV=7.8%) adult caribou. Total number of caribou (adults and yearlings) estimated on the calving ground in 2017 was 262,272 (SE=16,746) in June 2014 and 252,060 (SE=15,493). Weighted log-linear regression of the adult female estimates from 2008, 2014, and 2017 suggest a non-significant decreasing trend with a yearly λ estimate of 0.98 (CI=0.94-1.01) further indicating a longer-term declining trend of 2% (CI=-6% to +1%) per year. Using a regression simulation approach for the 2008, 2014, and 2017 surveys, Boulanger et al. (2018) were further able to demonstrate that the majority of trend estimates suggested a negative trend (λ<1). The mean λ estimate in this case was 0.975 (percentile 95% CI=0.95-1.00) which is similar to that obtained from regression analysis.

Community-based information has also raised considerable concern for the future of the herd across the Kivalliq region due to recent unpredictable movements of the herd across its spring, calving, and summer range in addition to a thriving inter-territorial meat sales market largely between the Kivalliq and Baffin Regional communities. These concerns were heightened with a documented drop in relative densities of calving Qamanirjuaq caribou between reconnaissance surveys flown in June 2008, 2010 and 2012.

Table 1. A survey history of the Qamanirjuaq Herd showing estimates and the methods used to derive estimates. Photographic survey methods provide the most reliable results and are generally used when relative densities are high.

Year	**Total Herd Size**			**Source**
	Y_h	**SE**	**CV**	
1968	63,000			Parker, 1972 *(Visual Calving-ground Survey)*
1976	43,800			*Calef & Hawkins, 1981* *(Visual Calving-ground Survey)*
1977	44,095	n/a	n/a	Heard, 1981 *(Visual Calving-ground Survey)*
1980	39,000	n/a	n/a	Heard & Calef, 1986 *(Visual Calving-ground Survey)*
1982	180,000	n/a	n/a	Heard & Calef, 1986; Gates, 1985 *(Visual Calving-ground Survey)*
1983	230,000	59,000	0.258	Heard and Jackson, 1990; Thomas, 1998; Williams, 1995 *(Calving-ground Photo-Survey)*
1985	272,000	142,000	0.523	Heard and Jackson, 1990; Thomas, 1998; Williams, 1995 *(Calving-ground Photo-Survey)*
1988	221,000	72,000	0.328	Heard and Jackson, 1990; Thomas, 1998; Williams, 1995 *(Calving-ground Photo-Survey)*
1994	495,665	105,426	0.213	Unpublished data; Thomas, 1998 (Calving-ground Photo-Survey)
2008	344,078	48,861	0.081	Campbell, Nishi, & Boulanger, 2010 *(Calving-ground Photo-Survey)*
2014	264,718	21,913	0.088	Campbell, Boulanger, & Lee. 2015 *(Calving-ground Photo-Survey)*
2017	288,244	22,438	0.078	Campbell, Boulanger, & Lee. 2018 *(Calving-ground Photo-Survey)*

The present work was designed to re-assess the abundance and trend of the Qamanirjuaq mainland migratory barren-ground caribou subpopulation. We designed the survey to meet the following 4 objectives: 1) Obtain an estimate for the number of females on the calving ground with a coefficient of variation of <15%; 2) Determine the trend in herd abundance since 2008; 3) Estimate the ratio of breeding females to the total number of females at peak of calving as an indicator of productivity; 4) Delineate the spatial extent of the annual calving ground and compare this to historical calving ground use.

2.0 STUDY AREA

Using annual location data collected from satellite and GPS collars between 1993 and 2013 we estimated the Qamanirjuaq caribou herd range to cover an estimated 310,000 km^2, (**Figure 1**). The annual range is large with its northern extents starting from the southern shores of Baker Lake and Chesterfield Inlet (latitude 57 degrees north), extending south to northeastern Saskatchewan and northern Manitoba. The Qamanirjuaq range is bounded to the east by the Hudson Bay coastline and to the west by longitude 105 degrees. The annual range covers four jurisdictions NWT, Manitoba (Man), Saskatchewan (Sask), and Nunavut (NU), and includes seven communities; Brochet Man., Tadoule Lake Man., Black Lake Sask., Wollaston Lake Sask., Arviat NU, Whale Cove NU, Rankin Inlet NU, Baker Lake NU, and Chesterfield Inlet, NU. Most of the annual range including the calving and post-calving range, as well as the spring and fall migration corridors, lie entirely within Nunavut, while the early- mid- and late-winter ranges extend into all four jurisdictions.

The Qamanirjuaq caribou annual range extends from the northern Arctic ecozone at its northeastern edge through the southern Arctic ecozone into its largest expanse in the taiga shield ecozone and ending with its southern tip within the boreal shield ecozone and at its southeastern tip within the Hudson plain ecozone (Environment Canada, 2009, **Figure 2**). Though the herd occupies five different ecozones, it rarely ranges into the northern arctic ecozone and are more commonly observed within the southern arctic ecozone during spring and summer. Dominant seasonal range within the southern arctic ecozone include spring migratory, calving, post-calving and portions of the early summer range. The Taiga Shield covers the largest portion of the Qamanirjuaq herds annual range making up most of the herds late summer, fall migration, fall rut, early winter, and late winter seasonal ranges while the Hudson Plains and Boreal shield are less commonly used across most years (Campbell et al. 2012: Environment Canada, 2001; Wiken, 1986).

The Southern and Northern arctic ecozones are dominated by open tundra largely made up of graminoid, herbaceous shrub, and ericaceous shrub habitats, with lichen mats common across aggregate glacial deposits, beach ridges, and rocky ridges. The Taiga Shield is dominated by open lichen woodlands with interspersed grasslands and shrubby habitats, While the Hudson plains and Boreal shield ecozones are more dominated by closed conifer and mixed forest, arboreal lichens, with interspersed grasslands and shrubby habitats.

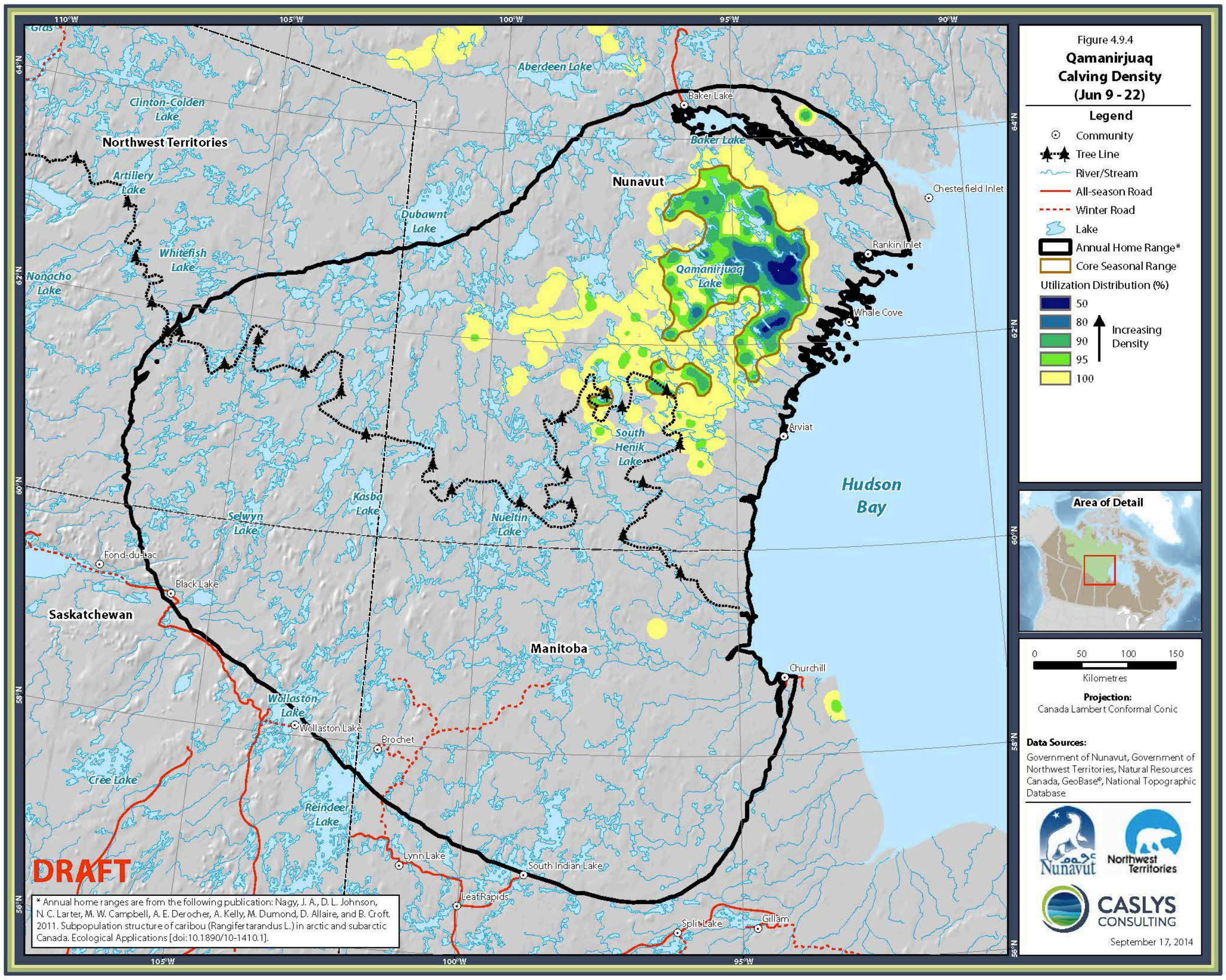

Figure 1. The annual range extents (bold black line) and annual concentrated calving area (dark blue, green, and yellow) of the Qamanirjuaq barren-ground caribou herd. Range extents were calculated using a kernel analysis of satellite and GPS collar data collected between November 1993 and April 2013 (Campbell et al. 2014).

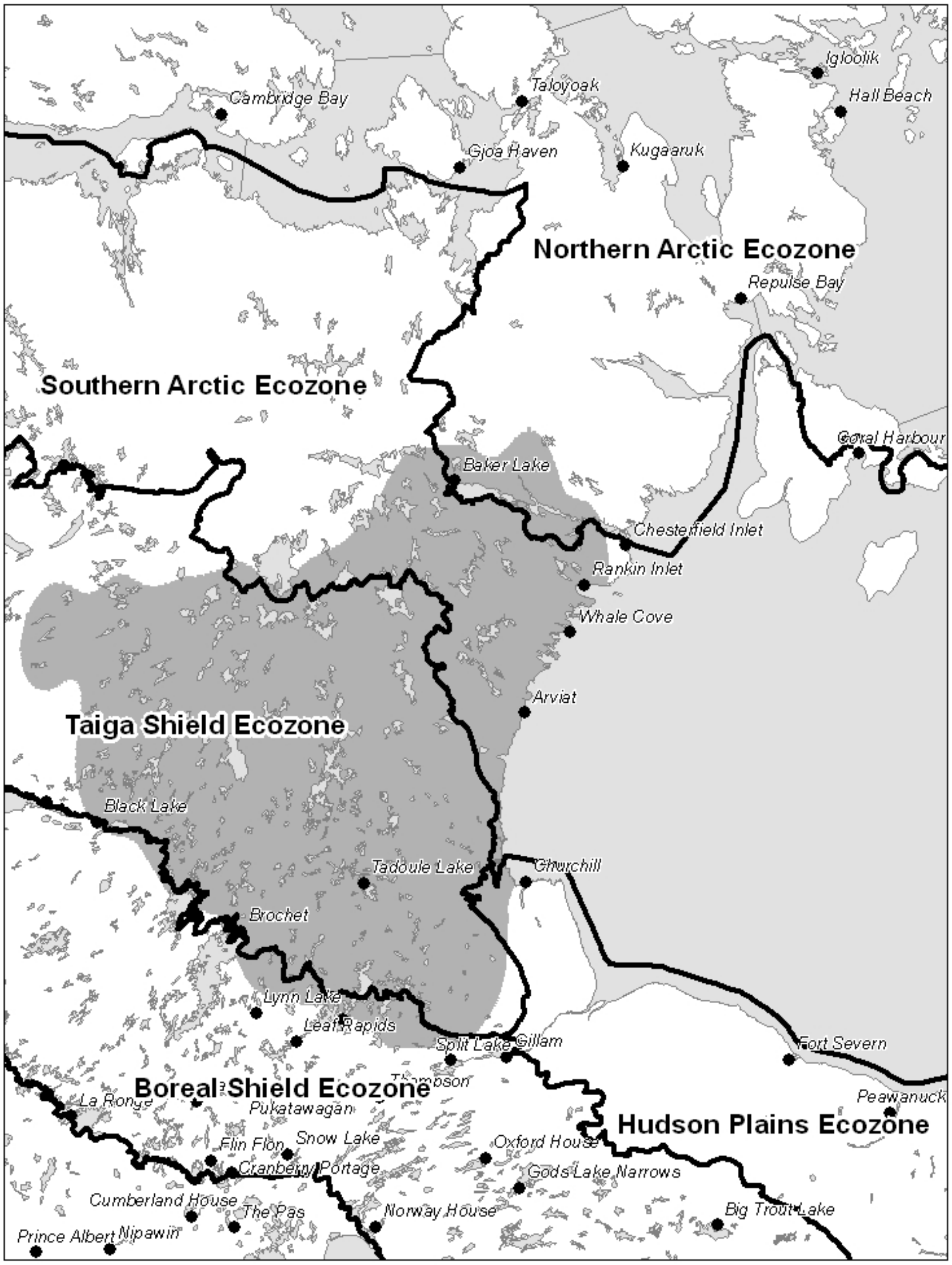

Figure 2 Ecozones of the Qamanirjuaq caribou herd annual range (1993 to 2008) (Environment Canada, 2001; Wiken, 1986). Dark shading is the known Qamanirjuaq annual range (The communities of Hall Beach and Repulse Bay have since been renamed Sanirajak and Naujaat respectively).

3.0 METHODS

The 2022 Qamanirjuaq barren-ground caribou double observer pair (DOP) visual and photographic calving ground survey was based out of the community of Rankin Inlet, Nunavut, with periodic refueling stops in the community of Arviat, 300 km south of Rankin. The survey was structured into five main components: 1) Systematic reconnaissance DOP visual survey, 2) DOP visual abundance survey, 3) Photographic abundance survey, 4) Density-based abundance strata composition surveys and 5) fall composition surveys. The systematic reconnaissance survey was initiated when GPS collar movement rates reached or fell below 5 km/day. The purpose of the reconnaissance survey was to develop strata to delineate discrete areas of similar breeding female and non-breeding female relative densities, commonly termed very low, low, medium and high. These delineated strata were then flown during the abundance phase of the survey using degrees of flying effort that would grade from high percent coverage in the highest density strata, to lower percent coverage within lower density strata. Aerial photography was utilized within the highest density strata where, in past surveys, densities have been found to be too high for accurate visual counts. The DOP visual abundance survey and the concurrent stratum-based composition surveys were used to estimate the total number of females and breeding females on the annual concentrated calving grounds, while fall composition survey results were used to extrapolate female abundance to whole herd estimates by incorporating the male to female ratio.

3.1 Visual Surveys

Two high-wing, twin engine, turbine, DeHavilland Twin Otter aircraft were used for both the reconnaissance and visual abundance surveys across the entire survey study area. Left and right observation strip widths were established using streamers attached to the wing struts (**Figure 3**). To configure the strip widths (w) on aircraft struts we used Norton-Griffiths (1978) formula:

$$w = W * h/H$$

Where:

W *= the required strip width;*

h *= the height of the observer's eye from the tarmac; and*

H *= the required flying height*

Based on decades of aerial wildlife survey work and analysis, the strip width was configured out to 400 meters at 122 meters above ground level (AGL) on each of the left and right aircraft struts, for a total transect width of 800 meters. All aircraft were equipped with radar altimeters to ensure an altitude of 122 meters AGL was maintained accurately while on transect. During the reconnaissance survey, caribou were classified where and when possible as adult with or without antlers, adult with or without calf, female, yearling, or bull.

The DOP method implemented during all phases of the June 2022 reconnaissance and abundance surveys is very similar to the strip transect method used in pre-2008 calving ground surveys with the exception of the addition of a second pair of observers. The double observer pair method allows for the comparison of caribou sightability between front observer pairs (primary observers) and rear observer pairs (secondary observers). Due to the high densities of observations typically encountered during barren-ground caribou surveys, we utilized a cooperative modification to the method between same side observers. This modification allows same side observers to discuss their observation as long as ample time is given to ensure each observer has actually seen the same group. For every observation, we collected additional information (co-variates) including visibility, ground speed, percent snow cover, elevation indices, and percent cloud cover to help estimate caribou sightability and resulting most representative statistical models used to assess abundance and associated reliability. Off-transect observations were optional during the abundance phase of the survey so that observers could maintain their primary focus on effectively searching the 400-meter strip of ground between the strut markers, also termed the observation bin.

To increase data entry speed without reducing accuracy, and to reduce the time required to perform preliminary analysis of reconnaissance data for abundance survey stratification, a digital data entry system, termed the “Aerial Wildlife Survey – Observation Collector” (AWS-OC), was utilized for this survey (Campbell et al. 2012). The software was developed by the Government of Nunavut, Wildlife Research Division, in collaboration with Integrated Ecological Research, Caslys Consulting Ltd, and Nunavut Tunngavik Inc (NTI), in 2011, and originally deployed on the June 2011 Beverly mainland migratory barren-ground caribou calving ground abundance survey (Campbell et al. 2012). Since its original launch, improved hardware, and some enhancements to the AWS-OC software had been undertaken prior to its deployment in June 2022 (Boulanger et al. 2018). The AWS-OC software operates with Windows editions 7 through 10 and was developed specifically for use in both independent and dependent double-observer pair aerial caribou surveys, including distance-sampling applications, to facilitate the collection of field data, and the subsequent management of the resultant observation dataset. This tablet-based system allows for the instantaneous entering of caribou group waypoints (observations) directly into a digital database. Data entry time was cut by approximately 50% over standard hand written datasheets, with the added benefits of continuous back up onto a USB drive into a digital database with no additional data entry required.

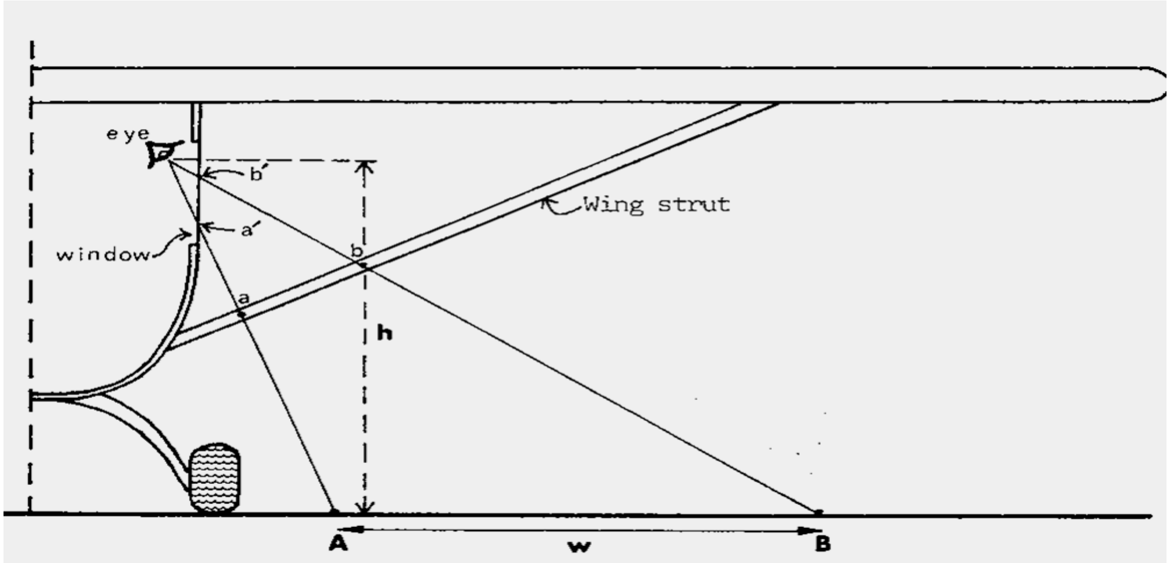

Figure 3 Schematic diagram of aircraft configuration for strip width sampling (Norton-Griffiths, 1978). W is marked out on the tarmac, and the two lines of sight a' – a – A and b' – b – B established. The streamers are attached to the struts at a and b, whereas a' and b' are the window marks.

3.1.1 Double Observer Pair (DOP) Visual Method.

The double-observer pair method was designed to increase the accuracy and precision of wildlife observations during visual surveys using both fixed wing and rotary wing aircraft. This method involves two pairs of observers on each of the left- and right-hand sides of the aircraft. Two "primary" or front observers who sit in the more forward seats over the wing struts and two "secondary" or rear observers who sit behind the primary observers (**Figure 4**). By design the method adheres to five basic steps:

1) The primary observer called out all groups of caribou (number of caribou and location) he/she saw within the 400-meter-wide strip transect before they passed halfway between the primary and secondary observer (approximately at the wing strut). This included caribou groups that were between approximately 12 and 3 o'clock for right side observers and 9 and 12 o'clock for left side observers (**Figure 4**). The main requirement was that the primary observer be given time to call out all caribou seen before the secondary observer called them out;

2) The secondary observer called out whether he/she saw the caribou that the first observer saw and observations of any additional caribou groups. The secondary observer waited to call out caribou until the group observed passed half way between observers (between 3 and 6 o'clock for right side observers and 6 and 9 o'clock for left side observer);

3) The observers discussed any differences in group counts to ensure that they had called out the same groups or different groups and to ensure accurate counts of larger groups;

4) The data recorder, one in front of the left side observers and the second in front of the right-side observers, categorized and recorded counts of each caribou group into "front only", "rear only", and "both", while recording predetermined co-variates; and

5) The left two observers and right two observers switched places approximately half way through each survey day (i.e. during refueling or within a stratum) to address

observer ability and sightability differences between the front and rear seats. The recorder noted the names of the front and rear observer for all observations.

The sample unit for the survey was "*groups of caribou*" not individual caribou. Recorders and observers were instructed to consider individuals to be those caribou that were observed independent of other individual caribou and/or groups of caribou. If sightings of individuals were within close proximity (within an estimated 200 meters) to other individuals then the caribou were considered a group.

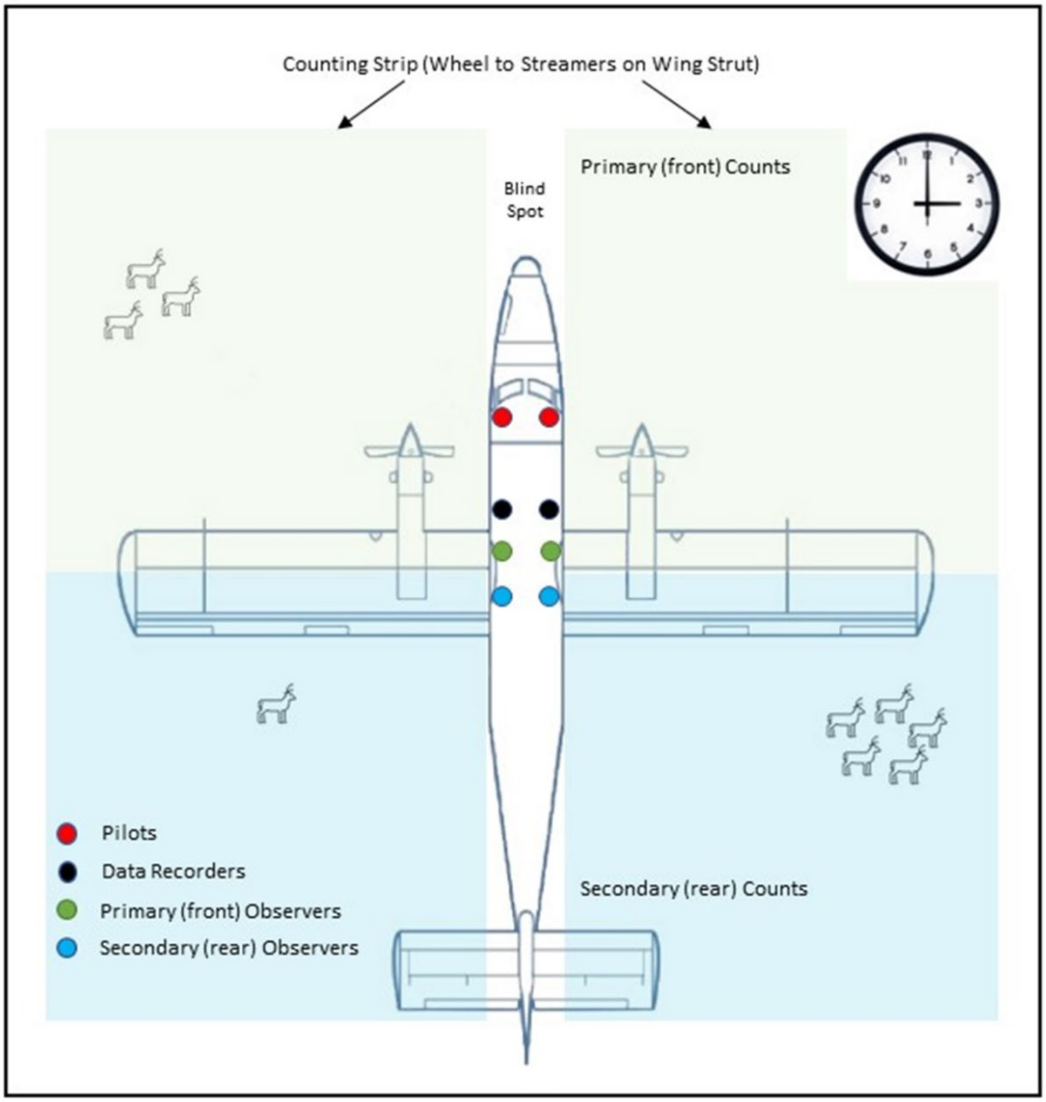

Figure 4. Observer position for the double observer pair method deployed on this survey using Twin Otter aircraft. The rear (secondary) observer (blue shaded area) calls caribou not seen by the front (primary) observer after the caribou have passed the main field of vision of the front observer (light green shading). The small hand on a clock is used to reference relative locations of caribou groups (e.g., “Caribou group at 3 o’clock” would suggest a caribou group 90° to the right of the aircrafts longitudinal axis.).

3.1.2 Systematic Reconnaissance Survey

The systematic reconnaissance survey was designed to estimate relative densities and delineate aggregations of females and breeding females (hard antlered cows or cow/calf pairs) for the purposes of stratifying the calving ground for the subsequent photo and visual abundance surveys. We used the observed locations of hard-antlered cows, newborn calves and aggregations of bulls and yearlings to delineate the spatial extent of the annual calving ground (Heard and Jackson, 1990; Thomas, 1998; Williams, 1995; Bergerud et al. 2008). The systematic reconnaissance survey of the annual calving ground was flown June 6th and 7th, 2022. Some additional reconnaissance was flown June 9th following the completion of the abundance phase and along known spring migratory corridors to ensure distributions of breeding females were not missed.

The reconnaissance survey was based on a systematic array of transects running north-south (**Figure 5**) and spaced at 10-kilometer intervals. Each transect was further divided into adjoining 10 kilometer transect segments, with each segment identified by a unique alpha-numeric code assigned to the transect station defining its northern end. The reconnaissance survey used these pre-determined transect segments (defined as one 10 km segment between two transect stations) to bin caribou observations for the purposes of calculating relative density within the segment. A rigid set of criteria governed when the 10 kilometer transect segments were flown. Criterion controlling when and where transect segments would be flown varied slightly across the calving distribution.

As the historic distribution of the Qamanirjuaq herd consistently displayed a distinct northern boundary along the leading edge of known migratory extents, while the southern, eastern, and western extents showed more inter-annual variability. For these reasons, rules controlling when to discontinue a transect varied between the northern extent of the distribution and that of the southern, eastern and western. Consecutive transect segments were flown north along the reconnaissance grid until no females and/or breeding females (hard antlered cows or cow/calf pairs) were observed within the ten-kilometer segment being flown. To further define the northern limit of breeding and non-breeding cows, parallel ten kilometer transect segments, adjacent to the additional northern segment, would also be flown. Along the more southerly "trailing edge" of the observed caribou distribution, the reconnaissance survey continued two full transect segments (including those segments directly east and west) beyond any surveyed segment where fewer than 2 breeding females/females were observed. On the western extents where caribou densities were in excess of 5 animals per ten kilometer transect segment and/or breeding female densities below 2 per transect segment, additional western transects would be flown at 20 km spacing between transects rather than ten, to increase area coverage and to ensure aggregations of breeding females/females were not missed. We intermittently continued the reconnaissance along known spring migratory corridors to ensure distributions of females/breeding females were not missed (**Figure 6**).

Following the systematic reconnaissance but prior to the initiation of the visual and photographic surveys, all tabulated observations were entered in to ESRI GIS software to calculate relative densities of breeding females using a tool utility. The relative density tools were built in ESRI's Model Builder (v9.1) utility and loaded into Arc Toolbox. The tools allowed us to calculate the relative density of observed caribou locations along the reconnaissance transects and associated transect segments and display these results on a map. We used vector-based analysis methods based on the following steps: 1) The survey transect segments were buffered by a user-specified width (i.e., 800 meters) yielding polygons that were 8 km^2 (i.e., 0.80 km wide x 10 km long); 2) The survey observations points were intersected with the derived buffer polygons; 3) The density was calculated for each polygon by dividing the number of 1+ year-old caribou by the area of the buffer polygon (#1+ year old caribou/km^2); 4) The relative density (#obs/km^2) is then thematically displayed on a map based on pre-defined classes or bins. The resulting graphics were then used to stratify the breeding female/female distributions into High, Medium, low, and very low-density strata to prepare for the abundance phase of the survey.

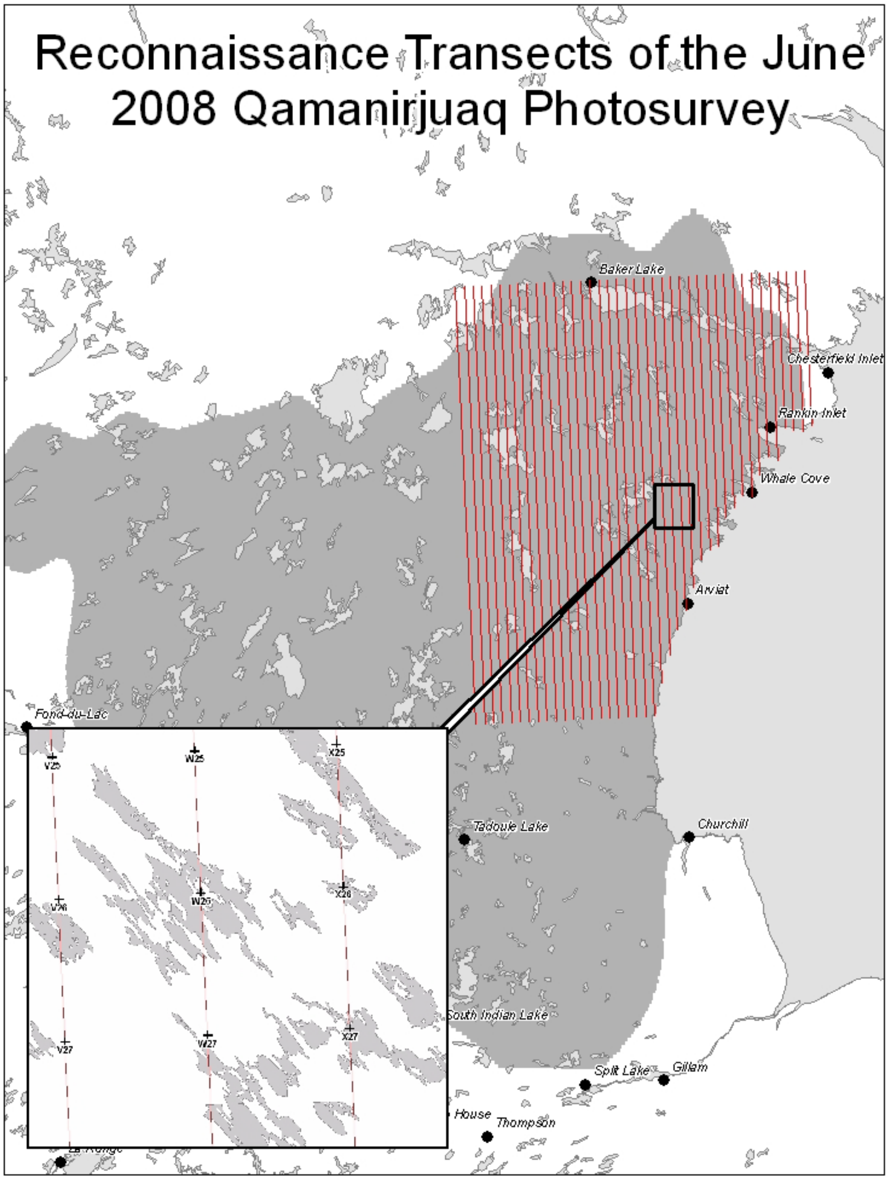

Figure 5. Potential reconnaissance transects and transect stations designed to cover the known extent of calving for the Qamanirjuaq barren-ground caribou herd in June 2022. These same transects were used in all consecutive surveys flown from June 2008 to present. Not all lines shown in this figure were flown during the 2022 survey.

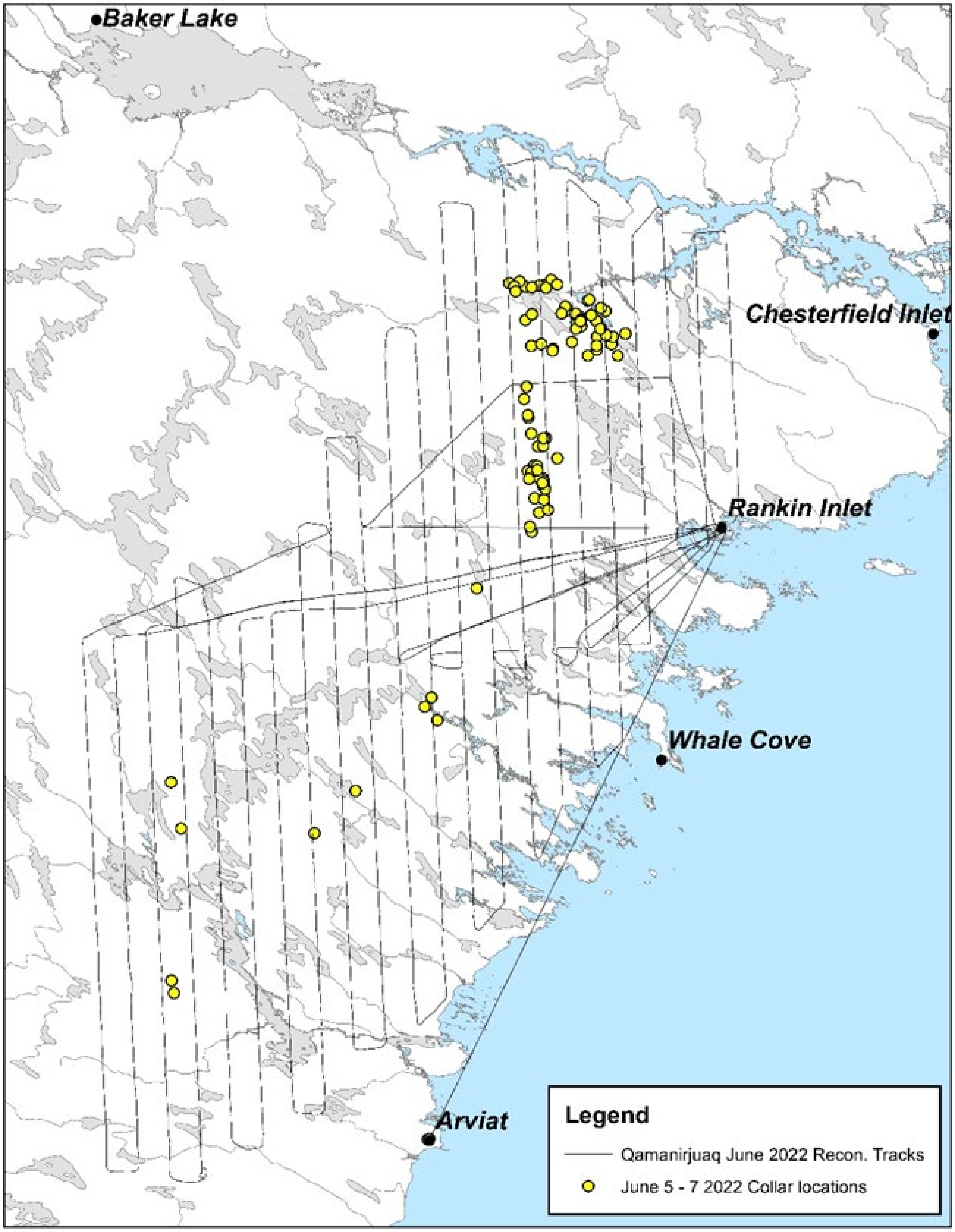

Figure 6. Qamanirjuaq Caribou Herd June calving-ground reconnaissance survey flown June 6th and 7th, 2022. Yellow dots show all collared Qamanirjuaq caribou cow locations over the same period.

3.1.3 Visual Abundance Surveys

The visual abundance survey was conducted within 4 low to medium density strata all located entirely within the breeding female/female distribution identified using reconnaissance survey results (**Figure 7**). Stratum boundaries were visually aligned with the relative density graphic to capture transect segments of similar density. All visual strata were surveyed immediately following the completion of the systematic reconnaissance of breeding female/female distributions.

The visual survey followed the same methods as the systematic reconnaissance survey with the exception of transect allocation, coverage, and alignment. Transects within each of one medium density (vis-1), one medium-low density (vis-4), and two low density (vis-3+4) strata were aligned at right angles to the longitudinal axis of the stratum to maximize the total number of transects (N). Transect spacing was allocated based on relative densities calculated within each individual stratum (**Figure 7**). Within the medium density stratum transects were placed three kilometers apart providing approximately 30% coverage, while within low and very-low density strata, transect spacing was set at 3.17 km and 10 km yielding 30% and 9% coverage respectively.

Visual survey data collected within each stratum were analyzed using Jolly’s Method 2 for unequal sample sizes (Jolly 1969 *in* Norton-Griffiths 1978). Only counts of adults were used for the final population estimates. Lake areas were not subtracted from the total area calculations used in density calculations.

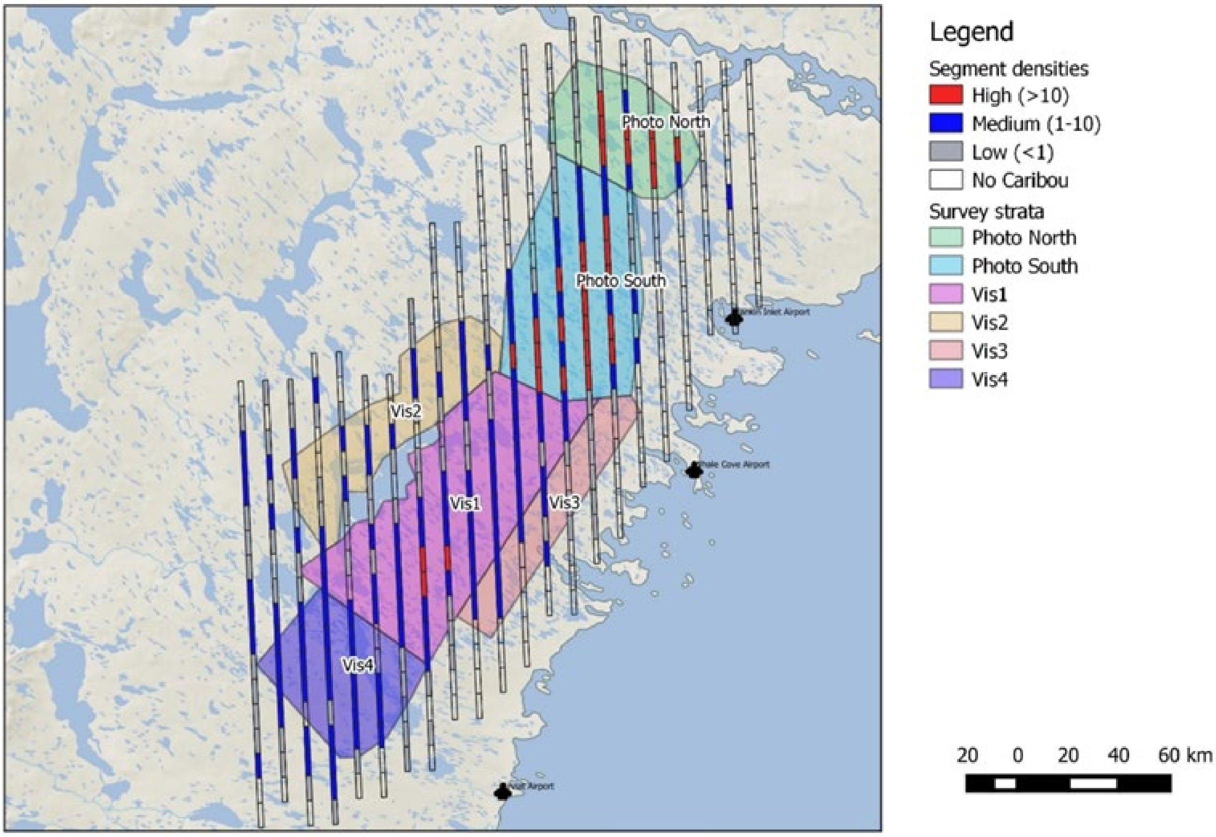

Figure 7. Reconnaissance transects with transect station relative densities overlaying strata derived using reconnaissance relative abundance segment estimates. Data collected, and strata derived during the 2022 Qamanirjuaq calving ground photographic abundance survey.

3.1.4 Photographic Abundance Surveys

Aerial photography provides more accurate abundance estimates of large mammals due to its ability to remove in-flight observer error and bias, and replace it with more controlled interpretation after the fact. This advantage is significantly increased when caribou relative densities exceed 10 to 15 caribou per kilometer squared, a point at which in flight observer error can become substantial. The photographic component of the survey used a single engine low wing Piper Malibu turbine aircraft. The aircraft was equipped with a radar altimeter and a digital camera with forward motion compensator.

The photographic component of the 2022 Qamanirjuaq calving ground survey was designed to capture relative densities of adult and yearling female and breeding female caribou in excess of ten caribou per kilometer squared as close to the completion of the systematic reconnaissance survey as possible. The photographic abundance survey was completed June 8th, 2022. As in the visual survey, transect spacing within the high-density photo strata was allocated based on proportional densities and available resources (Heard, 1987). During the June 2022 survey effort, high density transect coverage ranged from 45% to 54% coverage over the photo-north and photo-south photographic strata respectfully. Approximately 4,000 photos were taken within delineated photographic abundance strata, representing an estimated 900 linear kilometers of flying.

3.1.5 Double Observer Pair Visual Survey Analysis

Removal models in the mrds package were used to estimate and model sighting probabilities. In this context, double observer sampling can be considered a 2-sample mark-recapture trial in which some caribou are seen (“marked”) by the (“session 1”) primary (front) observer of which some are also seen by the secondary (rear) observer (“session 2”). The second observer may also see caribou that the first observer did not see. This process is analogous to mark-recapture except that caribou are sighted and resighted rather than marked and recaptured. A group of caribou rather than the individual caribou was the sample unit given that the sighting probabilities of caribou within a group were not independent.

In the context of dependent observer methods, the sighting probability of the secondary observer was not independent of the primary observer. To accommodate this, removal models were used which estimated *p* (the initial probability of sighting by the primary and secondary observer) and *c* (the probability of sighting by the secondary observer given that it had been already sighted by the primary observer). Note that resighting probability (*c*) is not equivalent to the initial sighting probability of a caribou (*p*). Also, the removal model assumed that the initial sighting probability of the primary and secondary observers was equal. Therefore, observers were switched midway in each survey day, and covariates were used to account for any differences that were caused by unequal sighting probabilities of primary and secondary observers (as discussed later). The combined probability that a group of caribou was seen by at least one of the observers (p^*) was therefore $1-(1-p)(1-p)$. **Figure 8** provides a conceptual argument for how p^* is estimated. It is p^* that is then used to estimate the overall sightability of caribou and adjust counts for caribou not sighted.

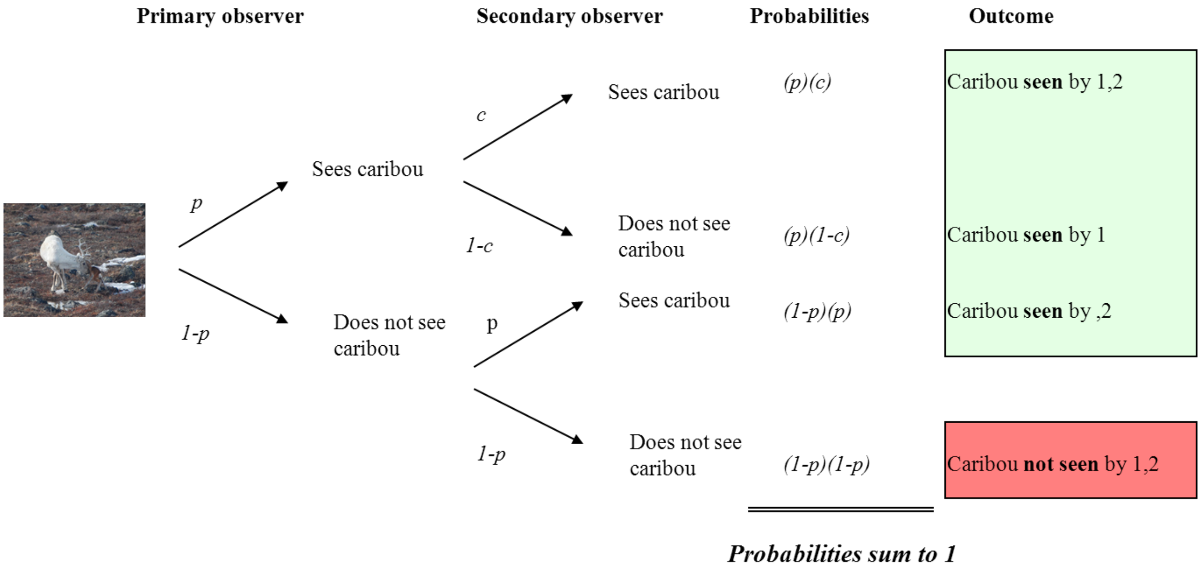

Figure 8. Conceptual diagram of how the probability of both observers not sighting a caribou group is estimated, and how the probability that at least one of the observers sees the caribou group (p^*) is estimated. The green boxes correspond to outcomes where caribou are seen and the red box corresponds where both observers do not see a caribou group. Mark-recapture methods are used to estimate sighting probabilities for the primary observer 1 and primary observer 2 (using data from when each observer is situated as the primary observer). Using these probabilities, the probability that a caribou is not seen can be estimated. In a method analogous to flipping a coin, each observer will see or not see a caribou as described by p (caribou seen) or $1-p$ (caribou not seen). Each of these outcomes can then be multiplied to obtain the probabilities for both observers combined. Because the two observers do communicate, the events are not independent and therefore the re-sighting probability of the secondary observer has to be adjusted (to c) using behavioral response removal models when the caribou was called out by the primary observer. However, since the probabilities sum to 1 it is possible to estimate the overall probability that the caribou group is sighted (p^*) as one minus the probability that none of the observers saw the caribou $(1-p_{ob1})(1-p_{ob2})$ (the red box) or by summing the probabilities in the green box.

Estimates of caribou within survey strata, and associated variance, were estimated using the mark-recapture distance sampling (MRDS) package (Laake et al. 2012) in program R program (R Development Core Team 2009). In MRDS, a full independence removal estimator which models sightability using only double observer information (Laake et al. 2008a, Laake et al. 2008b) was used therefore making it possible to derive double observer strip transect estimates. Strata-specific variance estimates were calculated using the formulas of (Innes et al. 2002) with the “S2” encounter rate estimator (Fewster et al. 2009). Estimates from MRDS were cross checked with strip transect estimates (that assume sightability=1) using the formulas of Jolly (1969) (Krebs, 1998). Data was explored graphically using the ggplot2 (Wickham, 2009) R package. GIS operations were conducted using the simple features (Pebesma, 2018) R package and QGIS software (QGIS Foundation 2020).

3.1.6 Modelling of sighting probability variation

One assumption of the double observer method is that each caribou group observed had an equal probability of being sighted. To account for differences in sightability we also considered the following sightability covariates in the MRDS analysis (**Table 2**). Each observer pair was assigned a binary individual covariate and models were introduced that tested whether each pair had a unique sighting probability. Previous analyses (Campbell et al. 2012, Boulanger et al. 2014a) suggested that the size of the group of caribou had a strong influence on sighting probabilities and therefore we considered linear and log-linear relationships between group size and sightability (**Table 2**). Cloud and snow cover, recorded by data recorders, were recorded as ordinal rankings as they changed across any given observation entry. We suspected that sightability was most likely lowest in mixed snow cover conditions and therefore we considered both categorical and linear models to describe variation in sightability caused by snow cover. Cloud cover, or the lack there of, could also influence sightability by causing glare, flat light, or variable lighting. We used the same basic strategy to model cloud cover variation and snow cover variation. Survey phase (reconnaissance or visual abundance survey) was also considered.

Table 2. Covariates used to model variation in sightability for double observer analysis.

Covariate	**Acronym**	**Description**
observer pair	obs	each unique observer pair
group size	size	size of caribou group observed
	Log(size)	Natural log of group size
snow cover	snowF	snow cover (0,25,75,100)
	snowc	continuous
cloud cover	cloudcat	cloud cover (0,10,25,75,100)
	cloud	continuous
Strata	Strata	Strata
Survey phase	Phase	Recon or visual

The fit of models was evaluated using the Akaike Information Criterion (AIC) index of model fit. The model with the lowest AIC_c score was considered the most parsimonious, thus minimizing estimate bias and optimizing precision (Burnham and Anderson, 1998). The difference in AIC_c values between the most supported model and other models (ΔAIC_c) was also used to evaluate the fit of models when their AIC_c scores were close. In general, any model with a ΔAIC_c score of less than 2 was worthy of consideration.

3.1.7 Data recorder observations

Data recorder observations, where data recorders saw caribou that were not observed by observers, were recorded for all of the observer pairs. Data recorder observations do not necessarily need to be included in analyses given that the method allows for observers to miss caribou and therefore the fact that a small percentage of caribou are only seen by data recorders is not surprising. In the context of the dependent double observer method, use of data recorder observations presents some challenges. First, observations from the data recorder are partial; the data recorder only records observations that he/she observes but are not observed by either other observer which limits the ability to use data recorder observers as a unique third observer. In this context, data recorder observations supplement the secondary observer in "testing" the primary observer.

One approach to include data recorder observations is to pool the secondary observer and data recorder as a single observer. The main potential issue caused by this approach is that it will increase the difference in detection probabilities between the primary and secondary observer regardless of observer position therefore violating the assumption of equal detection probabilities between observers. This could be thought of as always having one primary and 2 secondary observers that have a combined higher detection probability. Because the removal estimator considers observer order, this approach could potentially cause a negative bias in detection probabilities with a subsequent positive bias in abundance estimates. This scenario would likely correspond to cases when both observers have reasonable sighting probabilities. Another scenario, that likely occurred, was where both observers were weak and not including data recorder observations substantively reduced observations leading to a negative bias in estimates. In this case, observer probabilities are low and cannot be estimated using the double observer data alone. To detect this potential scenario, we estimated detection probabilities with and without data recorder observations under that rationale that these pairs could be identified by large differences in detection probabilities with data recorder observations included and excluded. In this case observations from these pairs were potentially included in the analysis with the secondary and data recorder observations pooled.

3.1.8 Analysis of trend

As an initial step estimates were compared using a t-test (Zar, 1996) with variances and degrees of freedom calculated using the formulas of (Gasaway et al. 1986). This comparison gave an initial indication of change in population size, but did not consider the survey interval between two surveys.

Estimates of trend were derived using ratios of estimates for pooled and post stratified estimates. A simulation approach that assumed log-normal distributions of estimates was used to test for significance between successive estimates as well as confidence limits on overall (gross) change and yearly change in estimates. Confidence limits were then derived based on the 2.5th and 97.5th percentile of the resulting distributions of gross change (GC) and annual change (with $\lambda = GC^{(1/\text{survey interval})}$).

Weighted regression analysis was also used to estimate trend from the time series of data (Brown and Rothery 1993). Each estimate was weighted by the inverse of its variance to account for unequal variances of surveys, and to give more weight to the more precise surveys.

3.2 Composition Surveys

3.2.1 Calving

Composition studies were conducted concurrently with the abundance phase photographic and visual surveys that immediately followed study area stratification. Caribou were classified as yearlings (>/= 1.0 but < 1.1 years of age termed 1+ years of age in this document), bulls, cows with calves (< one month old) and 0, 1, or 2 antlers, cows with udders and 0, 1, or 2 antlers, and udderless cows with 0, 1, or 2 antlers. Breeding cows were tallied as cows with calves, cows with udders, and udderless cows with antlers. Adult udderless cows with no antlers were classified as non-breeding females while all remaining adult females were classified as breeding females. All remaining caribou were classified as either yearlings or bulls. The proportion of breeding and non-breeding females was then determined using these categorizations. Bootstrap methods were used to obtain variance estimates for all strata. In this case, 1,000 resampling's of the data were used and the mean and standard deviation from resampling were used as point estimates with associated standard error as a proportion of breeding females, non-breeding females, calves, yearlings and bulls (Manly, 1997).

Composition survey effort was allocated as consistently as possible within each stratum. Selection of flight paths were based on fuel cache locations and caribou aggregations but consistently used the reconnaissance transect station locations in an attempt to maintain consistent coverage throughout the strata being sampled. GPS waypoints were recorded for all groups of caribou where they were first encountered.

June composition surveys were timed to begin concurrently with abundance photographic and visual surveys to ensure minimal movement between strata. Sampling was structured to begin at a fuel cache then proceeded to a predetermined transect station within a maximum of two (2) kilometers of the strata corner/boundary. From this station a Bell 206 Long Ranger aircraft would proceed to the next nearest transect station to the north and/or south, priority sampling the next nearest caribou group including individual caribou (**Figure 9**). At times, observed groups of caribou "pulled" the aircrew from the pre-planned flight path at which time the aircrew would stop sampling caribou groups that were seen greater than 10 kilometers (half the distance between reconnaissance transects) perpendicular to the original flight path. From this point, only caribou groups observed within this ten-kilometer buffer would be sampled and an attempt to rejoin the original flight path made. During re-positioning flights from the stratum to the fuel caches, caribou encountered within a maximum of 2 km inside of target stratum boundaries were classified opportunistically and variation of flight paths was held to within 2 km to conserve fuel and reduce deviation from the planned flight paths and fuel caches.

Estimates of the proportion of females and breeding females were then multiplied by the double observer pair estimate of all adult caribou and yearlings for each stratum to obtain an estimate of the number of non-breeding and breeding females within the survey study area extents. Variances were obtained for the combined estimate using the delta method (Seber, 1982; Williams et al., 2002) assuming no correlation between the two estimates.

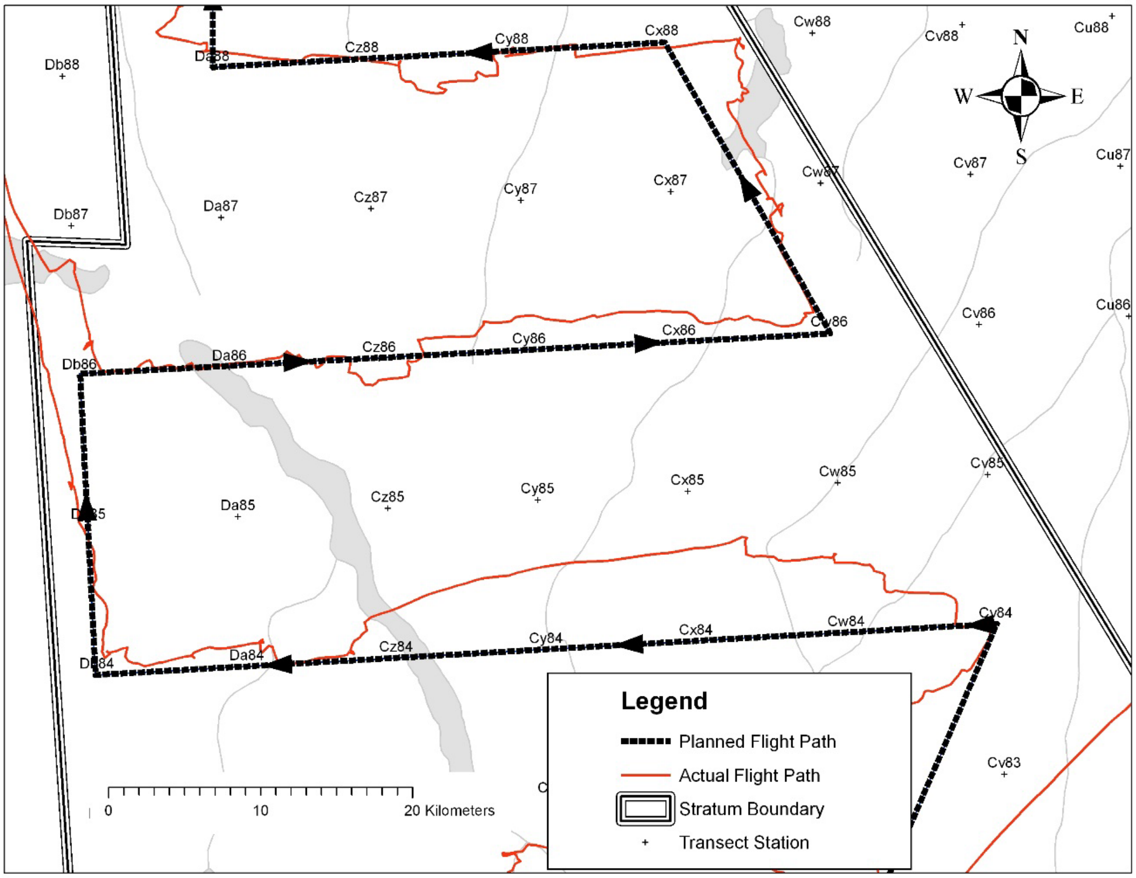

Figure 9. Strata composition flight lines vs. planned routes. Deviations were the result of observed caribou groups away from flight paths. The next nearest group would be classified up to a maximum of 10 km (half way between adjacent transects) perpendicular to the planned flight path.

3.2.2 Fall/Rut

The purpose of a Qamanirjuaq fall-rut composition survey is to determine the proportion of females in the population at a time of year when all age and sex classes come together into large mixed groups for the breeding season (rut). Though a combined estimate of breeding and non-breeding females, used in this report, is the most precise, and as such, represents the best indicator of population trend, for management and conservation education purposes, an estimate of total population size is desirable.

The Qamanirjuaq caribou fall composition survey was flown out of Arviat Nunavut, Tadoule Lake Manitoba, and Lac Brochet Manitoba, between October 15th and 20th 2014 (**Figure 10**). The survey itself used the locations of 20 Telonics GPS III and IV collars to locate aggregations of caribou and establish search patterns. All caribou groups encountered between and in the immediate vicinity of collared adult female caribou were classified and tracks followed to locate other groups. All collar locations were searched a minimum of twenty kilometers to the north, east, south and west of the outer most collar locations, with exceptions made when adjacent areas included boulder fields, large lakes, the Hudson Bay coast, or were restricted due to fuel limitations. Fresh tracks in snow were used in all areas to locate new groups. The search of an area represented by collared caribou cows would terminate following the cessation of fresh tracks or when a possibility of double sampling occurred. GPS tracks were also used to ensure the same groups were not re-sampled, which at times required the skipping of groups where there was a risk of mixing. Once the area around a collar or cluster of collars was thoroughly searched, the survey would proceed to the next nearest collar to begin a similar search pattern. In total, 121 groups, or 8,856 individual Qamanirjuaq caribou were classified.

To estimate the total population size, the number of non-breeding and breeding females estimated in June 2022, was divided by the product of the proportion of females in the population as determined during the 2014 fall composition studies. The proportion of females in the population assumed a 50:50 sex ratio for yearlings. We suggest that the proportion of females estimated on the calving ground is a better estimator of herd size and trend as the proportion of females pregnant, used to extrapolate a whole herd estimate from breeding females alone, is based on dated information and for the Qamanirjuaq population, not sampled since fall 2014. It should be noted that inter-annual variation in sex ratio does occur though large changes in herd sex ratio would be considered a-typical for these large mainland migratory herds (Gunn et al. 2005; Sober, 1982).

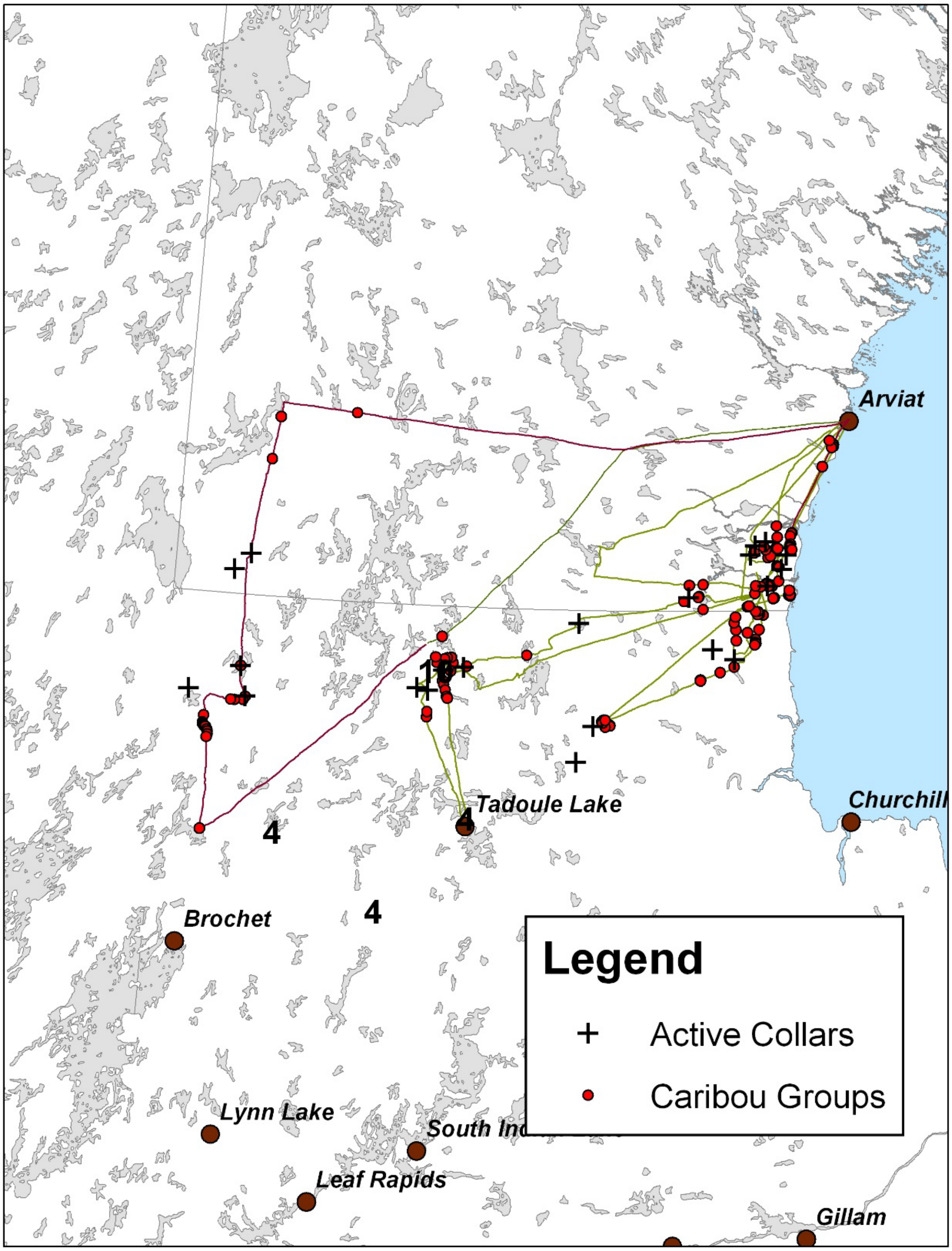

Figure 10. Qamanirjuaq fall composition flight tracks and caribou group locations, October 15 – 20th, 2014 (numbers indicate alternate fuel caches).

3.3 Spatial studies

Utilization distributions were generated for the calving season using telemetry data collected for Qamanirjuaq caribou between 2000 and 2022. Data were resampled to daily locations to address differences in sampling frequencies between collars and only locations collected between April 15th and July 3rd were considered in the analysis. All telemetry locations were attributed with a season based on the following date ranges: spring migration (April 15 – June 8), calving (June 9 – June 22), and post-calving (June 23 – July 3rd). To account for variation in the calving dates between individuals and years, each collar was reviewed and the start and end dates for the calving season were adjusted based on changes in movement pattern. Once the calving date ranges had been refined, a kernel density estimation (KDE) approach was used to generate utilization distributions for a series of data subsets to illustrate changes in calving distributions for Qamanirjuaq through time. The first calving time series examined changes in distribution over four years: 2008, 2014, 2017, and 2022. For each of the four years, the calving data were extracted and used to generate a KDE using a fixed bandwidth value of 9.5 km. The fixed bandwidth value is the average of the individual bandwidths calculated for each year using Silverman (1986). A fixed bandwidth was selected over the individual values to balance the differences in sample sizes between the four time periods. The second time series examined changes in distributions between five-year periods: 2003-2007, 2008-2021,2013-2017, and 2018-2022. As with the single year time series, KDEs were generated for each five-year subset using a fixed bandwidth of 12.6 km. The final time series examines annual changes in calving distributions using KDEs generated for each year from 2004 to 2022. As there were fewer than 30 locations for the calving season for years 2000 to 2003, the locations for these years were pooled to provide a more representative sample of calving distributions for this period. A fixed bandwidth of 12km was used to generate the annual kernel density utilization distributions.

4.0 RESULTS

4.1 Layout of Survey Strata

Survey strata were designed based on reconnaissance survey flight density observations, and the movements of 44 collared female caribou. The threshold for the peak of calving was based upon observations of cow/calf proportions and/or when movement rates declined to less than 5 km per day for collared cows (**Figure 11**).

The Qamanirjuaq June 2022 abundance survey was the shortest on record, largely due to the relatively small geographic area within which the spring migration occurred, geographically tight aggregations of females on the calving ground, and excellent weather. The survey took a total of 5 days to complete, 2 days for the reconnaissance survey, one day for the photographic survey, two days for the visual abundance survey, and 3 days for the composition survey (**Table 3).**

Survey strata were laid out based on reconnaissance survey segment densities (**Figure 12**) and the initial fixed wing composition of caribou observed within reconnaissance segments (**Figure 13**). Unlike June 2017, the main migration path of collared caribou cows occurred within a narrow corridor inland from, but parallel to, the Hudson Bay coast, as initially determined using adult cow collar locations, later verified during the reconnaissance survey (**Figure 14**). High densities of adult female caribou were observed within the northern extents of the 2022 Qamanirjuaq herd annual concentrated calving area which were stratified into photographic strata. Further south of these higher female densities were the visual strata, mainly composed of non-breeding females, yearlings, and bull caribou. As the majority of observations and their associated densities fell below the 10 caribou/km^2 threshold, photographic methods were not required. Further to reconnaissance observations, the movements of collared caribou indicated that the caribou moved to the northwest in the photo stratum then circled back to the northeast. The photo north stratum was extended to the northwest to capture this movement as well as guard against movement to the northwest which occurred during the 2017 photo survey (**Figure 15**). The lines in this area were considered a buffer and preliminary with the potential of eliminating them if movement did not occur.

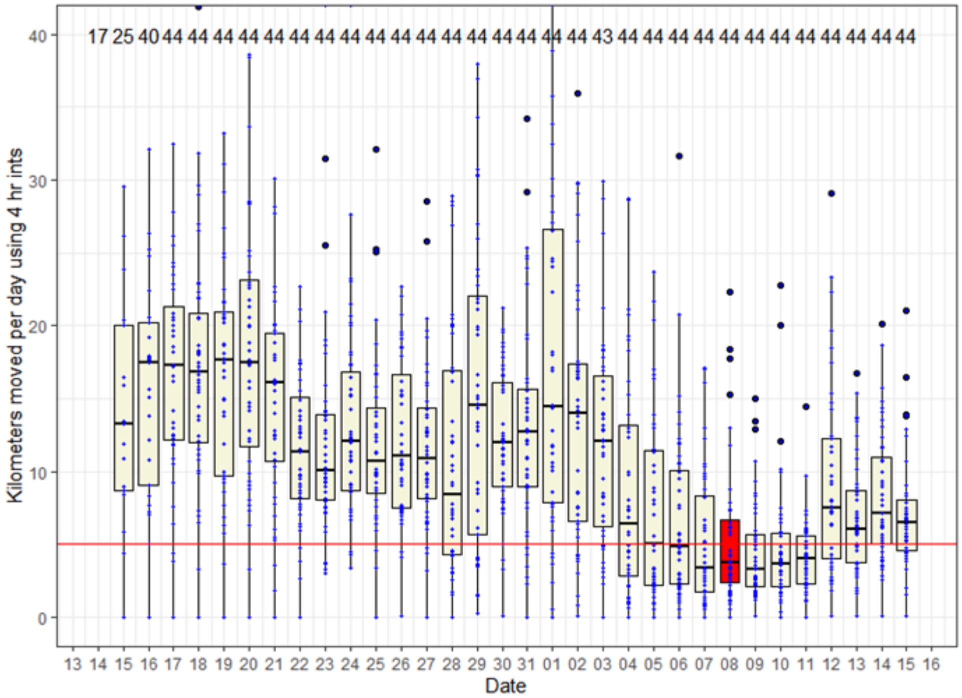

Figure 11. Collar movement rates of Qamanirjuaq cows during the June 2022 Qamanirjuaq calving ground photo survey. Sample sizes of collars are given above each boxplot. The boxplot for the primary date that the photo survey occurred is highlighted in red.

Table 3. Survey Initiation and completion dates for the June 2022 Qamanirjuaq Calving Ground Photographic Survey.

Survey Activity	Jun-06	Jun-07	Jun-08	Jun-09	Jun-10
Systematic Reconnaissance	X	X			
Visual Abundance			X	X	
Photographic Abundance			X		
Abundance Strata Composition			X	X	X

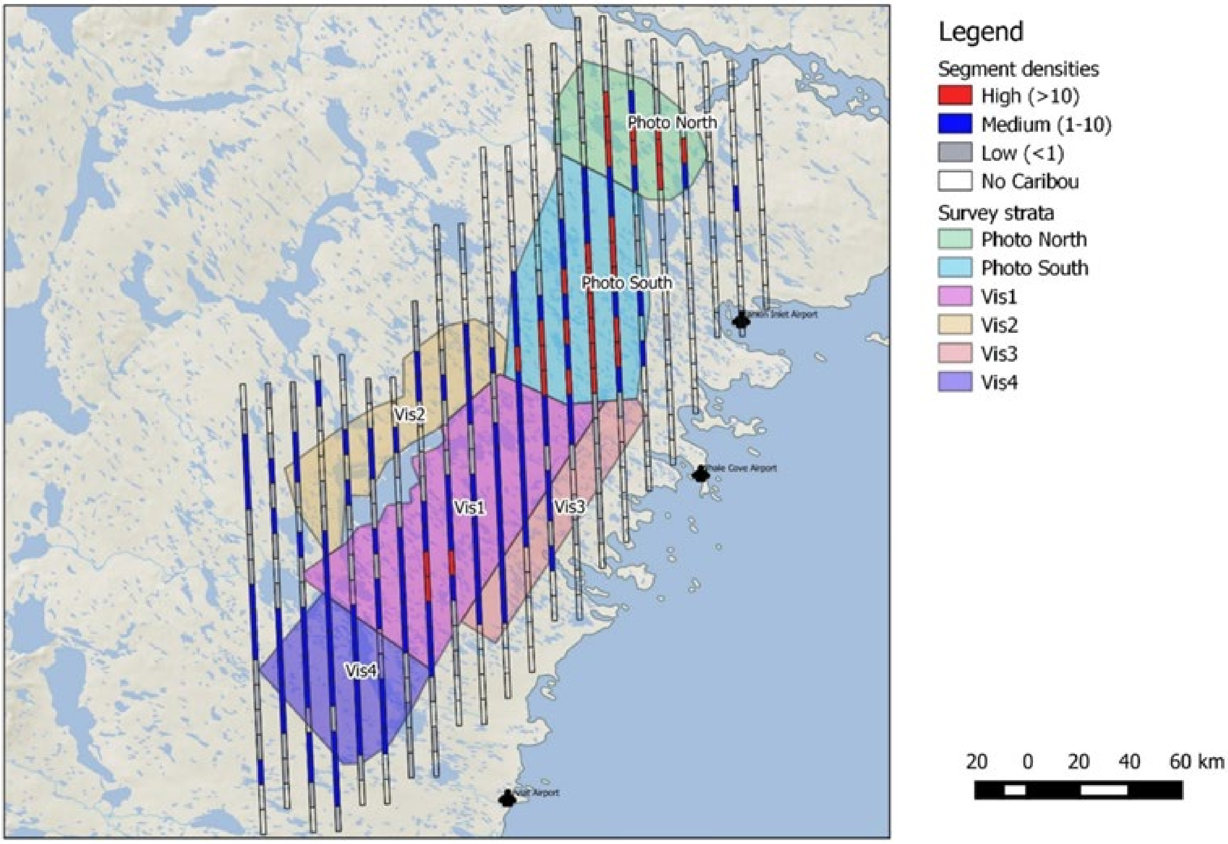

Figure 12. June 2022 Qamanirjuaq reconnaissance survey segment density.

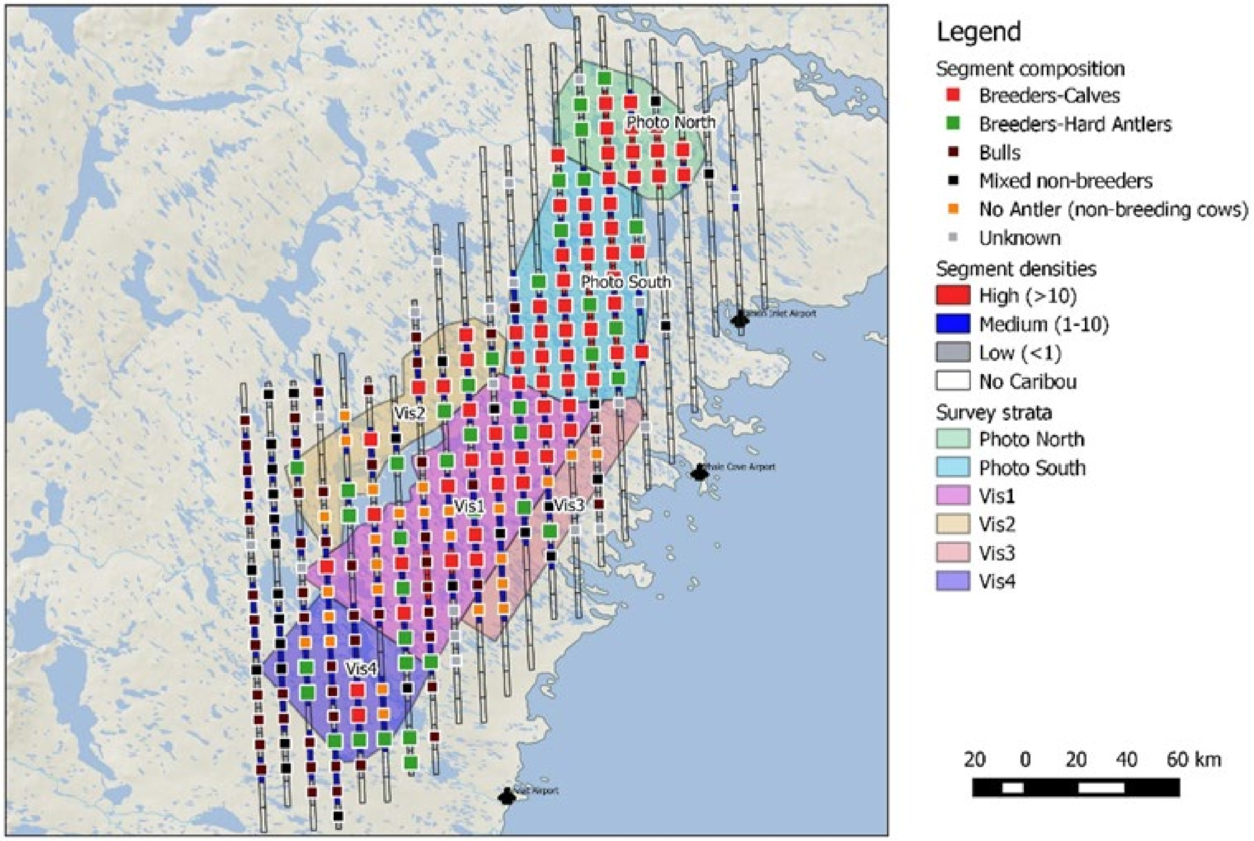

Figure 13. June 2022 Qamanirjuaq reconnaissance survey segment composition overlaid on segment densities as shown in Figure 12.

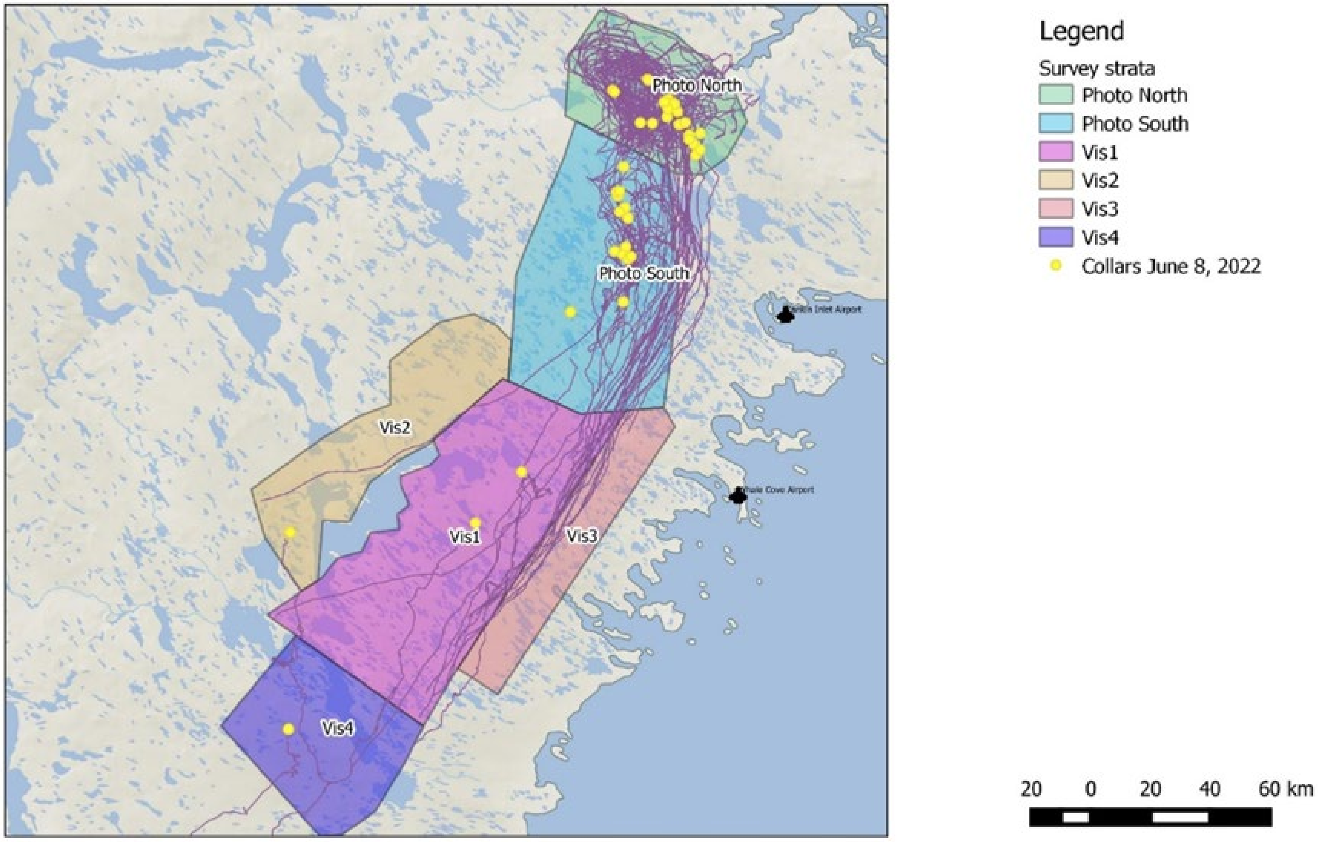

Figure 14. Qamanirjuaq June 2022 calving-ground survey collar movements from May 15 to the beginning of aerial survey observations June 8, 2022 (yellow points).

4.2 Allocation of Survey Effort

Following stratification, preliminary estimates of density were derived for each stratum which were then used to allocate the number of transects flown per strata (**Table 4**). Allocation for the photo strata was based on a limit of 5,000 to 6,000 photos total. Given that survey weather was favorable, we set a target GSD (Ground Surface Density or Ground Sampling Distance) level of 7 which would be equivalent to an estimated 5,000 photos to complete all photographic strata (**Figure 15**). A GSD 7 would provide individual photographs within which the distance between two consecutive pixel centers, as measured on the ground, would be 7 cm. We also set an approximate limit of 1,600 km of flying on transect which amounts to slightly over a day of flying for a single photo plane. Allocations suggested higher coverage for the Photo-north stratum given that the mean density was higher than the Photo-south stratum, however, the Photo-south stratum was approximately twice the size of the Photo-north strata and therefore received more transect km's of flying.

Table 4 The June 2022 Qamanirjuaq calving-ground abundance survey transect allocations based on reconnaissance observations and achieved GSD levels within photographic (high density) strata.

Strata	**area**	**Recon estimates**			**Allocation (GSD 7)**				**Actual flown**		
					Transects						
		Density	N	CV	SE-based	N-Based	% effort	Coverage	Transects	Km	Coverage
North	2121.3	12.9	27,470	64.0%	23	24	45.5%	47.1	19	664	44%
South	4131.8	7.4	30,502	35.5%	17	16	54.5%	22.7	20	958	33%
										1,622	

Graphically, it can be seen that the desired resolution, coverage, and photo numbers would be achieved at either a GSD 7 or 8, with GSD 8 more desirable as it would require fewer photographs (**Figure 15**). Given a stable high-pressure system in the survey area we judged these targets were reasonable.

The remaining strata were surveyed visually with allocations based upon the total number of kilometers that the two survey planes could fly in two days of flying assuming two trips per day with ferrying to survey strata factored into the calculations (**Table 5**). This amounted to 3,000 kilometers of flying on transect (including ferrying in-between transects). The Vis-1 strata had the highest densities of caribou outside of the photographic strata and likely the highest proportion of both breeding and non-breeding female caribou within non-photographic (Visual) strata and therefore it received higher coverage than other visual stratum. The Vis-4 stratum had higher densities, however, it was composed of mainly non-breeders and therefore received slightly lower coverage than visual 1.

Table 5. Allocations of visual strata based on 3,000 km's of flying on transect.

Strata	area	Recon estimates			Allocation					Actual flown	
		Density	N	CV	SE-based	N-based	%Effort	Coverage SE	Coverage N	transects	km
Vis1	5496.8	3.4	18,906	14.2%	39	33	72.4%	28.2%	24.0%	35	1,611
Vis2	2570.9	1.5	3,797	28.6%	23	21	13.6%	16.6%	15.0%	20	487
Vis3	1713.0	1.2	2,130	36.2%	20	22	6.5%	14.6%	15.6%	20	311
Vis4	2453.8	2.3	5,691	11.0%	5	12	7.5%	7.0%	17.4%	12	519
											2,928

Figure 15. Strata coverage and number of photos estimated at different aircraft altitudes during the June 2022 Qamanirjuaq calving-ground photo-survey.

4.3 Survey Layout Used for Estimates

The Photo-north stratum was expanded west to buffer for potential west movement of caribou, and to capture the migration path of caribou to their annual core calving area (**Figure 16**). Inspection of the distribution of caribou on the photos revealed that caribou had moved east out of the western extents of the stratum that were initially set up as a western buffer. These lines contained few (65) caribou on transect and were therefore not representative of this high-density photographic stratum. As a result, the western most transects were removed to account for this movement within the photo-north stratum (**Figure 16**).

Table 5 summarizes the dimensions and sampling effort for each of the strata sampled. The area surveyed in each stratum was estimated by the total transect kilometers flown multiplied by the strip width of the survey (0.8 km for visual and with variable widths for photo stratum). Coverage was estimated as the area surveyed divided by the strata area. Naïve density for stratum was then estimated as the total count of caribou divided by the area surveyed. From this, it can be seen that the density of caribou on the high-photographic strata were much higher than the visual stratum.

A preliminary estimate of abundance can be gained by dividing the caribou counted by coverage (**Table 6**). This estimate is preliminary for visual surveys given that estimates are not corrected using double observer methods. However, the preliminary estimate demonstrates that the actual means of obtaining strata estimates is relatively simple. It is just the estimate of caribou counted divided by the proportion of each strata sampled (the coverage). A plot of visual and photo survey results (**Figure 17**) suggests that the high-north photo stratum delineated the core group of female caribou as defined by caribou counted on photos as well as satellite collar locations and proportions. The migration trail was then sampled by the photo south and visual transects.

Table 6. Summary of sampling and count-based results by strata. Preliminary N is estimated based on the number of caribou counted on transect divided by survey coverage.

Strata	Strata area (km^2)	Transects	Area surveyed	coverage	Caribou counted	Density on transect	Preliminary N
Photo strata							
Photo North	1585.0	15	664.0	41.9%	41,314	62.22	98,601
Photo South	4390.4	20	1383.4	31.5%	24,945	18.03	79,190
Visual strata							
Vis1	5496.8	35	1288.5	23.4%	5,323	4.13	22,748
Vis2	2570.9	20	389.4	15.1%	435	1.12	2,881
Vis3	1713.0	20	248.8	14.5%	425	1.71	2,931
Vis4	2453.8	12	415.5	16.9%	1,066	2.57	6,308

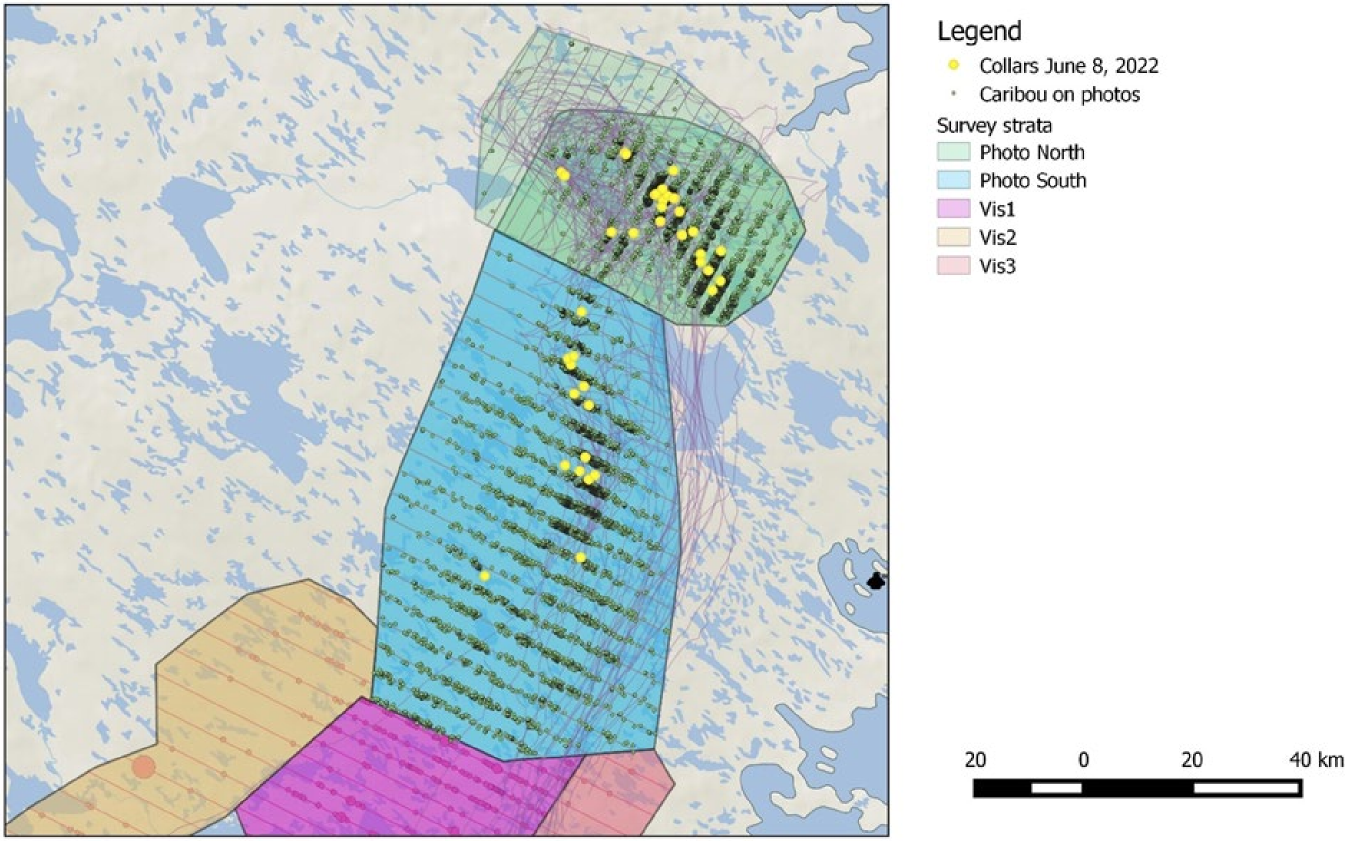

Figure 16. The 2022 Qamanirjuaq June calving-ground survey reduced photo north stratum (darker green area) in comparison to full photo-north stratum (Light green area).

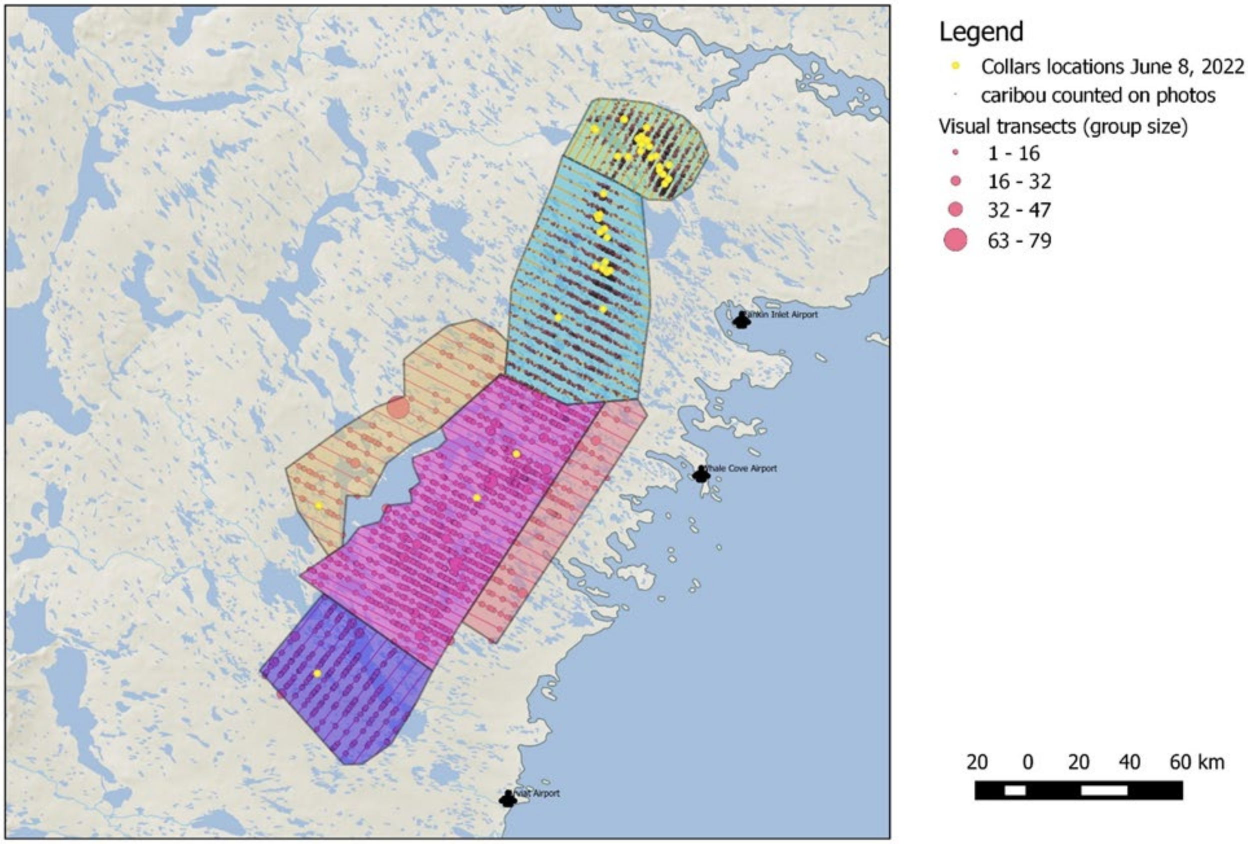

Figure 17. Summary of photo and visual survey with group sizes indicated for visual surveys and densities of individual caribou shown for photo data (light green and blue areas), for the 2022 Qamanirjuaq June calving-ground survey.

4.4 Analysis of Survey Data

4.4.1 Visual survey double observer pair surveys

The majority of caribou were seen as single caribou or small groups with few larger group sizes observed. The relative proportion of caribou not seen by both observers was highest in group sizes of 3 or less with both observers seeing the majority of group sizes that were greater than 3. Compared to previous surveys (Campbell et al. 2012), the proportion of caribou seen by both observers was high suggesting that overall sightability was high (**Figure 18**).

During the reconnaissance surveys the core of the calving ground was surveyed which led to observations of larger group sizes (**Figure 19**). This area was surveyed using the photo plane for the abundance phase of the survey and therefore the number of larger groups was lower during the abundance phase. The main focus of the analysis was to estimate sightability for the visual survey phase. For this reason, the reconnaissance survey data set was filtered to only include group sizes of 80 or less caribou which was similar to the range of group sizes observed in the visual survey. Photo strata counts increased the precision of group counts within high density aggregations, and visual strata had significantly lower densities and associated smaller group sizes. Additionally, the visual counting of areas of very-high caribou densities generally leads to a breakdown in establishing the defined 200-meter separation between groups as observers often combine groups due to the longer time it takes to count more caribou. Because of this, large groups often flow into one another yielding higher group sizes. Because the main focus of the analysis was to estimate sightability for the visual survey phase, the recon survey data set was filtered to only include group sizes of 80 or less which was similar to the range of group sizes observed in the visual survey.

Overall, there were 6 unique pairs of observers during the visual portion of the survey. Of these pairs, 5 of them switched position from primary to secondary during the survey (**Table 7**). One pair (pair 0) only occurred during the reconnaissance survey and did not switch and for this reason was not used in the analysis. Pair 2 was composed of 4 individuals given that the 2nd pairing only had 12 sightings preventing modelling of sighting probabilities. Data recorder observations mainly occurred for pair 1.

Table 7. Summary of observer pairings used in the double observer analysis.

Pair	Observers		Observation type/frequencies				
	front	rear	Both	Front	Rear	Data recorder	Total
0	JR	OS	63	11	0	0	74
1	DM	OS	413	15	125	60	613
1	OS	DM	82	6	0	16	104
2	DL	JV	441	31	9	8	489
2	JV	DL	104	5	25	5	139
2	DL	LI	9	0	0	0	9
2	LI	DL	0	2	1	0	3
3	JB	LI	574	7	14	8	603
3	LI	JB	227	5	13	2	247
4	JE	RT	80	20	7	0	107
4	RT	JE	486	13	47	1	547

A graphical representation of detections suggests most detection differences occurred when group sizes were low (**Figure 18**). Pair 1 had a higher relative frequency of missed observations than other pairs, however, some differences also existed between other pairs **(Figure 19).**

The pooled data from observer pairs (**Table 8**) suggested slight differences in proportions of caribou sighted as indicated by the proportion of caribou only observed by the secondary observer. Graphically, it can be seen that there is minimal difference between detection probabilities when data recorder observations are included except for pair 1 **(Figure 20).**

Though the reconnaissance survey phase saw variable cloud conditions, in general, survey conditions were ideal with 0% cloud during the abundance phase (**Figure 21**). Proportion of missed observations was slightly higher when cloud cover was 50% or greater during the recon survey phase, though consideration must also be given to observer experience, which would increase with every successive flying day. Snow cover was consistent for both the reconnaissance and abundance phases of the survey with over 80% of observations recording 0% snow cover for the reconnaissance, and over 95% of observations recording within 5% to 25% snow cover for the abundance.

Table 8. Summary of double observer pairings with sample sizes and naïve detection probabilities for each pair. Observations are summarized by observation type (BO-both observers, DR-data recorder, FO-front observer, RO-rear observer). Naïve detection probabilities are based upon proportions not seen by the front observer. Single (p) and double observer (p2x) probabilities are shown. They are calculated excluding data recorder observations (no DR) and including data recorder observations (DR).

Pair	Observations (type)				Totals	Naïve detection probabilities			
	BO	FO	RO	DR	total	P1x (no DR)	p2x (no DR)	P1x (DR)	P2x (DR)
1	495	21	125	76	719	0.80	0.96	0.72	0.92
2	557	38	35	13	647	0.94	1.00	0.93	0.99
3	802	12	27	10	857	0.97	1.00	0.96	1.00
4	569	33	54	1	665	0.92	0.99	0.92	0.99

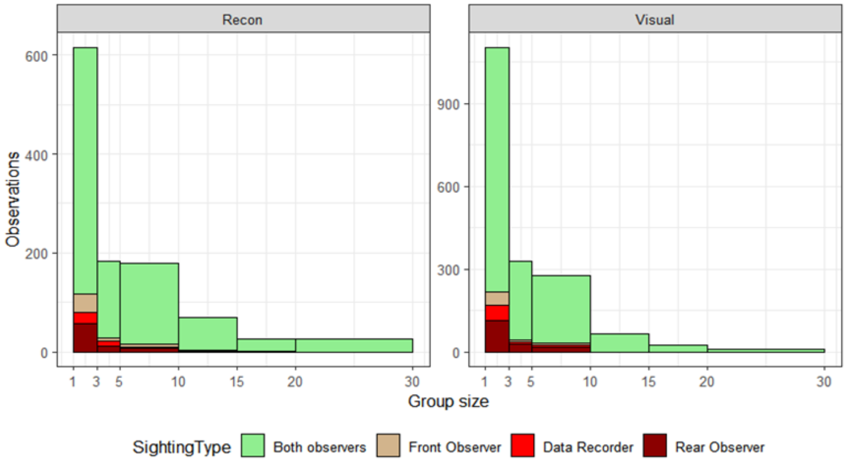

Figure 18. Distribution of group sizes observed during the visual and reconnaissance surveys (green) with observation type delineated as sub-bars (brown and reds). Group size observations of greater than 80 caribou (only observed during the reconnaissance survey) were not used in the double observer analysis. Observations greater than 30 caribou are summarized in the 20-30 bin.

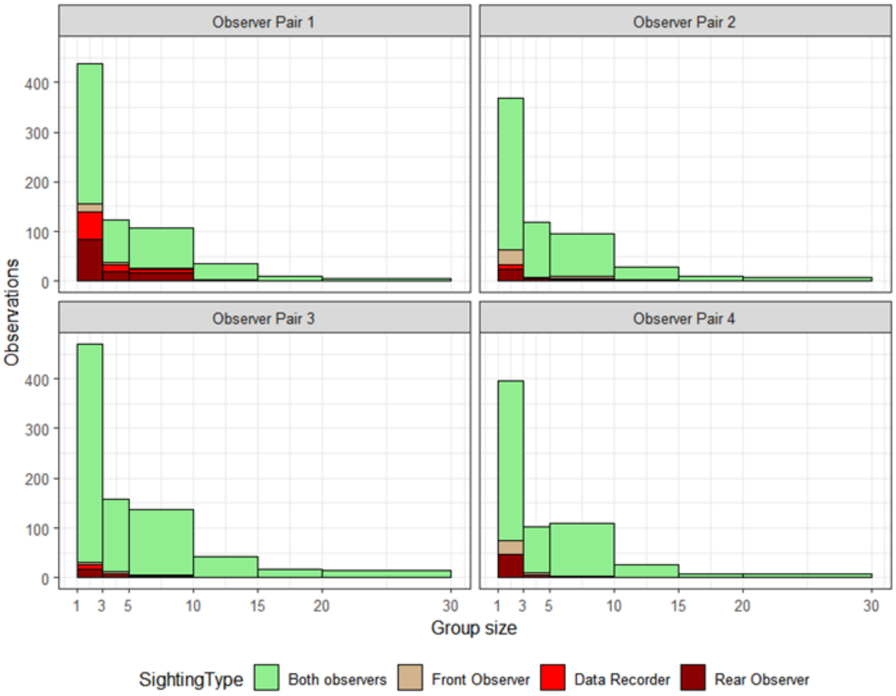

Figure 19. Graphical summary of observer pair detections by group size for the June 2022 Qamanirjuaq caribou calving-ground abundance survey.

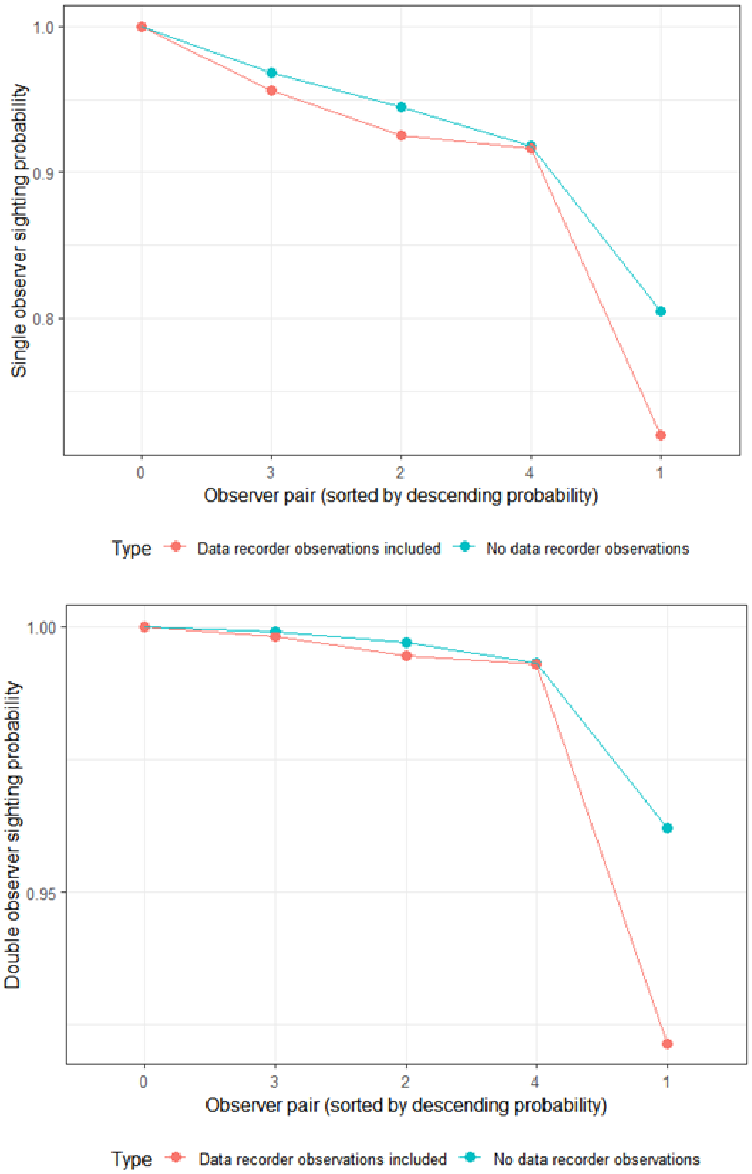

Figure 20. Graphical representation of the effect of inclusion of data recorder observations for the June 2022 Qamanirjuaq caribou calving-ground abundance survey.

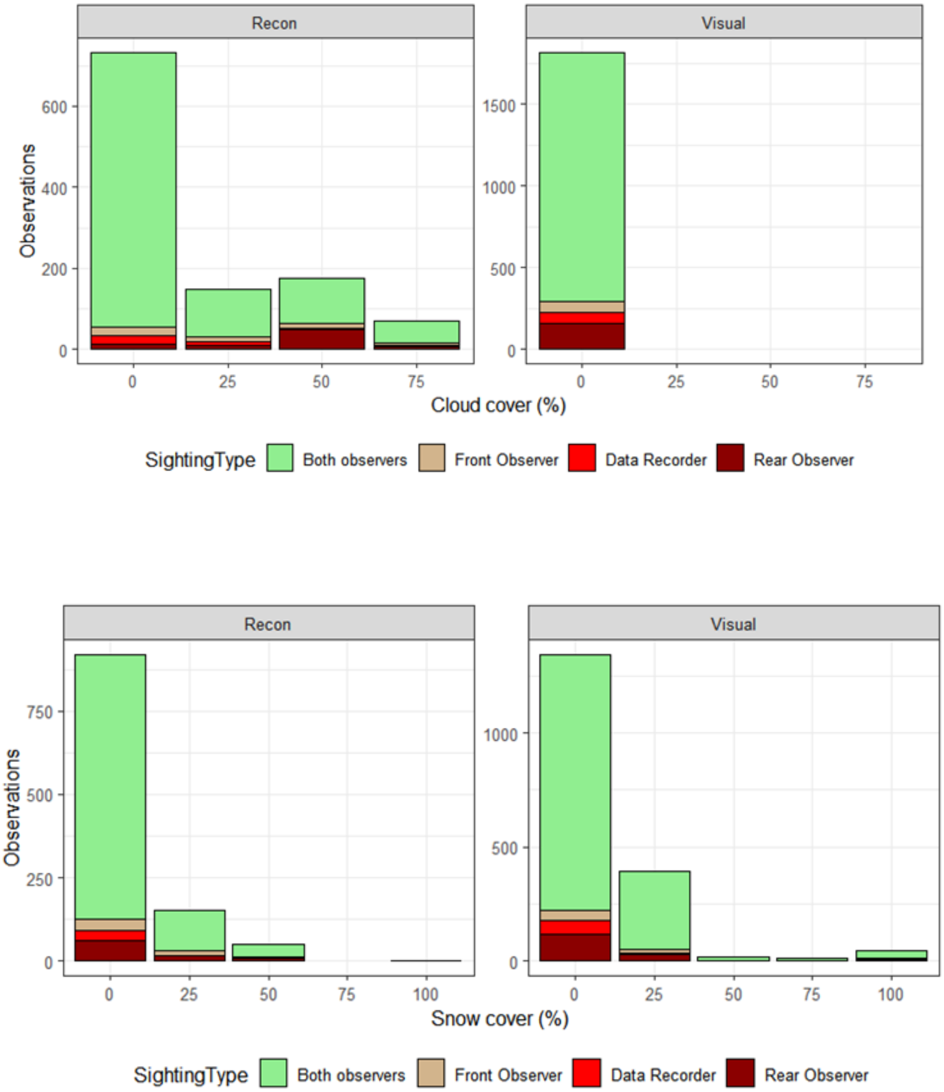

Figure 21. Summary of cloud and snow cover for observations during the recon and visual components of the survey.

4.4.2 Model selection

The general model building procedure followed a hierarchical process (**Table 9**). Initially, model building focused on determining the relative strength of each covariate with unique observer, group size, and cloud cover, showing substantially higher support than a constant model. Observer and group size were considered as additive and interactive terms, with interaction models showing higher support (model 6). Cloud cover was then considered as an additive continuous or categorical term. From this the most supported model (model 1) contained an interaction of observer (pair) and group size with an additive effect of cloud cover.

Table 9. Double observer model selection results. Main model terms are listed as columns with covariate names as defined in Table 1. Sample size adjusted Akaike Information Criterion (AICc), the difference in AICc between the most supported model for each model (ΔAIC_c), AIC_c weight (w_i), number of model parameters (K) and deviance is given.

No	Model	AIC_c	ΔAIC_c	w_i	K	Deviance
1	observer*size+cloudf	1790.53	0.00	1.00	11	-883.6
2	observer*size+cloudc	1807.89	17.35	0.00	9	-894.5
3	observer*size	1837.39	46.86	0.00	8	-910.4
4	observer*size+phase	1837.97	47.44	0.00	9	-909.6
5	observer*size+strata	1838.65	48.12	0.00	12	-906.6
6	observer+size	1840.96	50.43	0.00	5	-915.3
7	observer+size+phase	1841.36	50.83	0.00	6	-914.5
8	observer*log(size)	1842.62	52.09	0.00	8	-913.0
9	observer+cloudc+snowc	1868.98	78.45	0.00	6	-928.3
10	observer+cloudc	1869.73	79.20	0.00	5	-929.7
11	observer+snowc	1896.55	106.02	0.00	5	-943.1
12	observer	1897.38	106.85	0.00	4	-944.6
13	observer+phase	1899.20	108.66	0.00	5	-944.5
14	cloudf	2016.00	225.47	0.00	4	-1003.9
15	size	2031.77	241.24	0.00	2	-1013.9
16	log(size)	2036.57	246.04	0.00	2	-1016.3
17	cloudc	2053.82	263.29	0.00	2	-1024.9
18	strata	2092.80	302.27	0.00	5	-1041.3
19	snowF	2093.59	303.06	0.00	5	-1041.7
20	constant	2096.99	306.45	0.00	1	-1047.5
21	phase	2097.32	306.79	0.00	2	-1046.6
22	snowc	2098.44	307.91	0.00	2	-1047.2

The influence of observer pair, group size, and cloud cover on sighting probabilities (**Figure 24**) suggested that the largest degree of variation was due to group size and different observer pairs, however, the overall range in probabilities was not large. Furthermore, double observer probabilities (the combined probability of at least one observer in a pair sighting a caribou group) was close to 1 regardless of observer pairing or cloud cover (**Figure 22**). Lower detection probabilities did occur when cloud cover was 50% or greater, however, this only occurred during the recon survey and therefore had no effect on the visual survey estimates.

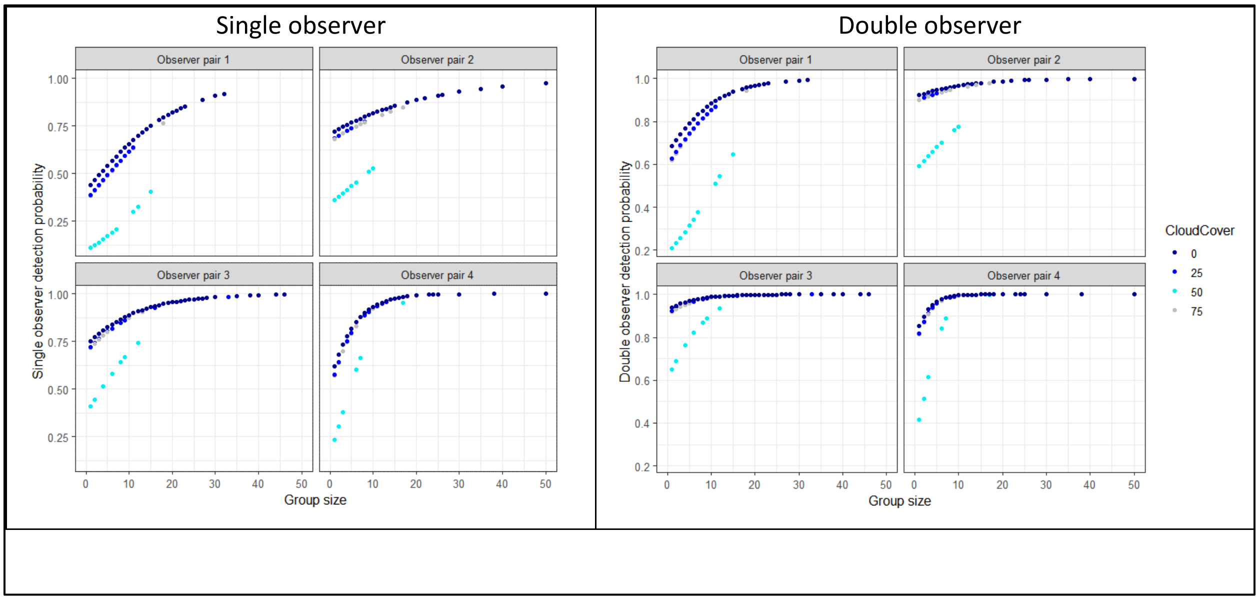

Figure 22. The Qamanirjuaq June 2022 calving-ground visual abundance survey single (left) and Double (right) observer sighting probabilities as a function observer and cloud cover.

Double observer estimates were derived in program MRDS from Model 1 (**Table 10**) and compared to non-corrected count-based estimates. In general, the estimates were very close (1.3% difference) with the total estimate for all strata being 473 caribou higher than the non-corrected estimate. As discussed later, the minimal difference in estimates was due to the larger group sizes encountered during survey (with high sightabilities (**Figure 22**)), and good survey conditions. Precision of double observer estimates was slightly higher due to the advanced methods used to estimate variance in the *mrds* package. Overall precision of estimates was quite high demonstrating that survey allocation and strata layout was optimal for obtaining precise estimates based on reasonable' km's of flying on transect.

Table 10. Double observer estimates of all caribou in each stratum and uncorrected count-based estimates for comparison purposes.

Strata	Caribou	Double observer (MRDS)				Count-based estimate			
	counted	N	SE	95% CI		CV	N	SE	CV
Vis1	5,323	23,083	1640.9	19,886	26,795	7.1%	22,708	1633.6	7.2%
Vis2	435	2,880	597.9	1,823	4,552	20.8%	2,872	506.0	17.6%
Vis3	425	2,996	351.2	2,310	3,886	11.7%	2,926	458.2	15.7%
Vis4	1,066	6,313	401.1	5,405	7,374	6.4%	6,295	868.7	13.8%
Total	7,249	35,273	1826.7	31,717	39,228	5.2%	34,801	1972.2	5.7%

4.4.3 High density photo and visual survey estimates

High density photographic strata were flown at GSD 8 which resulted in an average strip width of 1.35 km (sd=0.025, min=1.28, max=1.39, n=39). Strip width and transect area was measured using geo-refenced photos for each survey line. Transect densities were estimated as the number of caribou counted on a given transect divided by the transect area (**Figure 23**). Densities were above 10 caribou per km^2 in the High-density Photo-North Photo on all transects except two (2) on the west-central end of the stratum, and one (1) on far east end of the stratum. Very-high densities occurred on the central lines of this stratum with very-low densities on peripheral lines especially to the west. This variation reduced overall estimate precision.

Within the Photo-south stratum most densities were at the 10 caribou per km^2 level with densities up to 50 caribou per km^2 in the central section of the stratum. For the remaining visual abundance survey strata density of caribou along transects was below 10 caribou per km^2 (**Figure 24**).

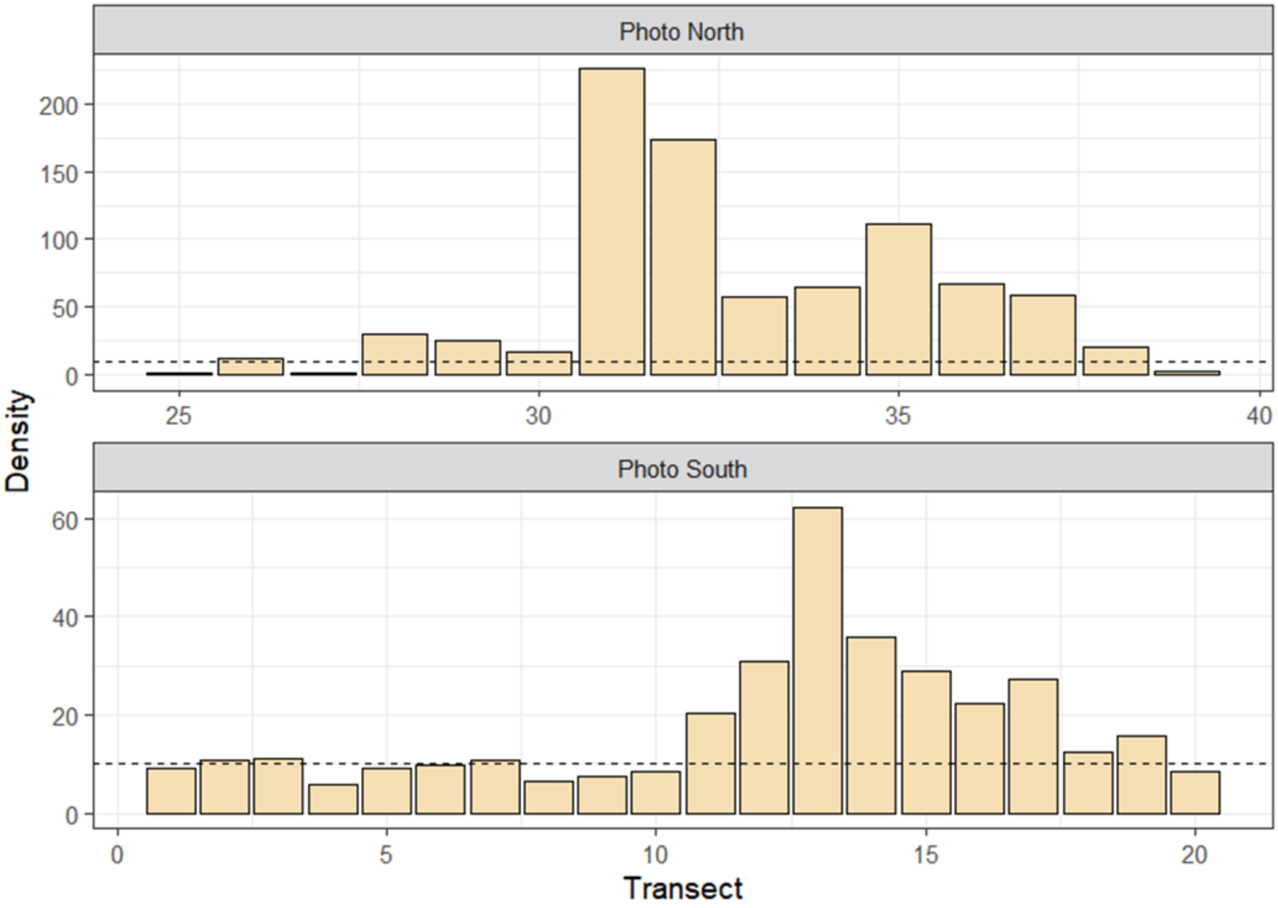

Figure 23. Estimates of caribou density (caribou per km^2) on high density photo stratum by transect. Note the different y-axis scales. Density of 10 caribou per km^2 which denotes the level when photo plane sampling is used is given as a horizontal dashed line. Transects went from west to east for the Photo-North stratum and from south to north for the Photo-South stratum.

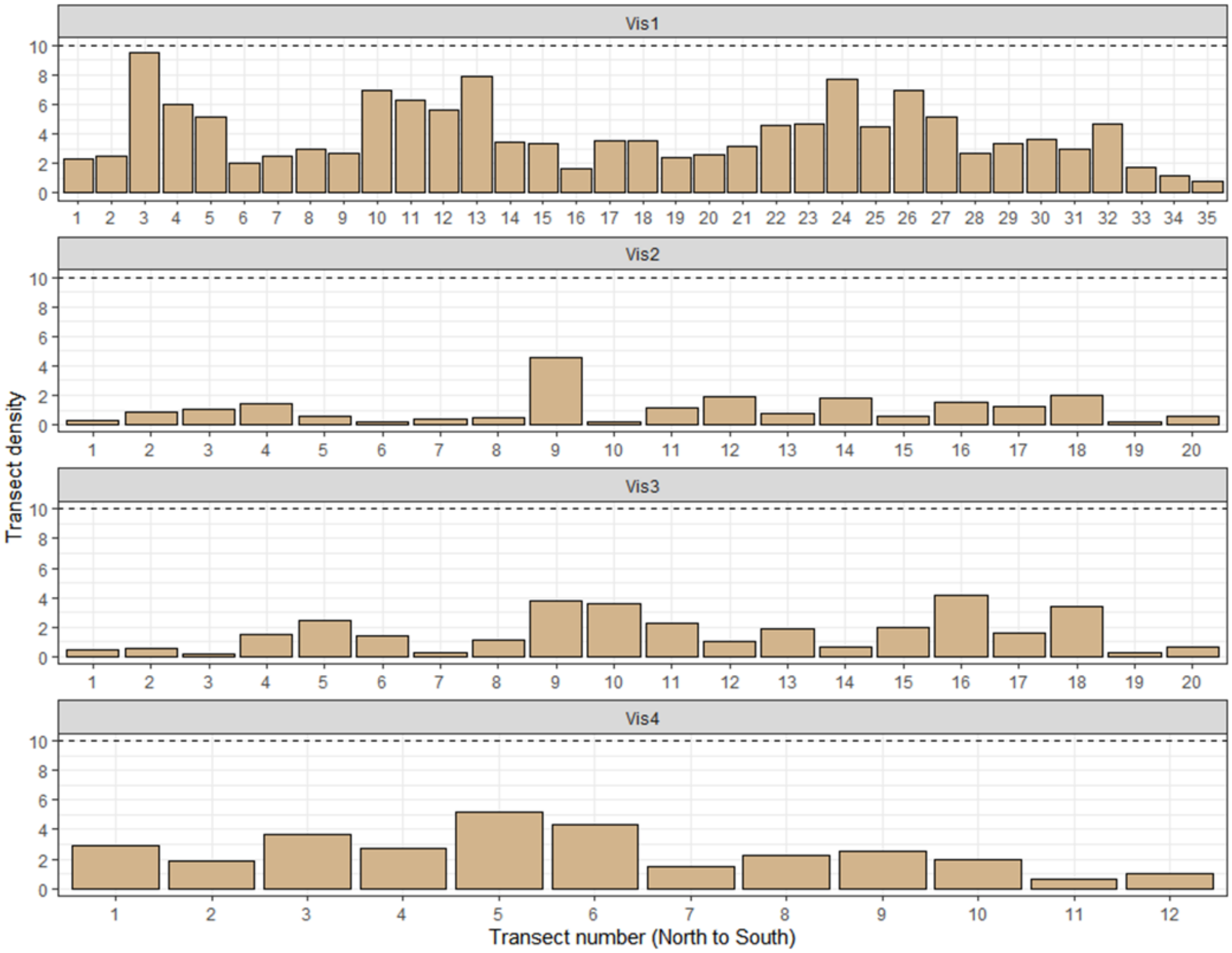

Figure 24. Transect densities within visual strata for the 2022 Qamanirjuaq calving-ground survey. The horizontal dashed line represents a density of 10 caribou per km^2 which denotes the level when photo plane sampling is used.

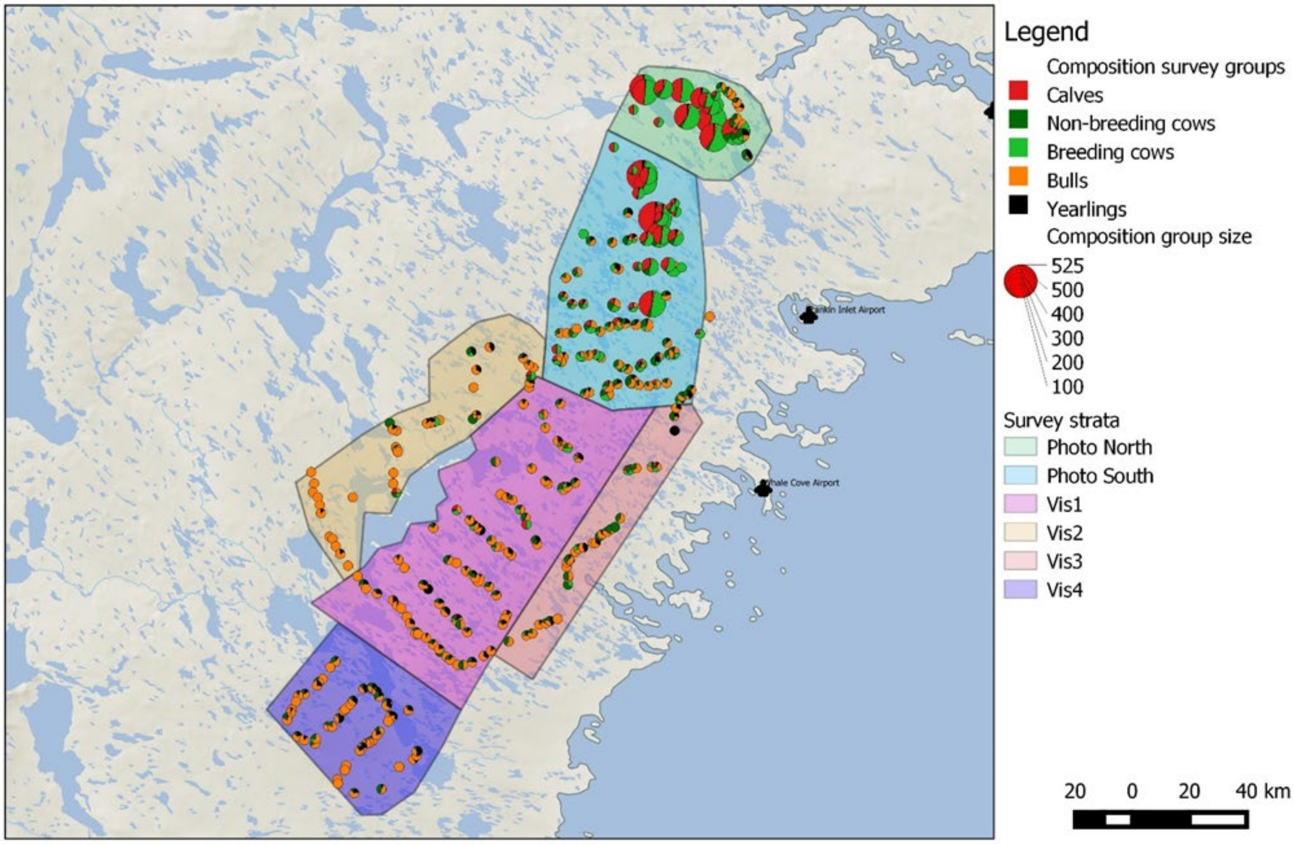

Figure 25. Summary of caribou classified for each of the core strata as listed in **Table 8** for the June 2022 Qamanirjuaq herd calving-ground abundance survey.

4.4.4 Composition surveys to determine proportions of females

Composition surveys were conducted on each of the photographic and visual survey strata (**Table 11**). A spatial representation of the composition data reveals that the majority of breeder groups including breeding and non-breeding females, occurred in the two photo strata with proportionally higher non-breeder/female groups occurring in the visual stratum as well as in the southern half of the photo south stratum (**Figure 25**). The composition data for all strata was analyzed further using a bootstrap procedure to estimate standard errors. One thousand bootstrap replications were conducted which resulted in robust standard error estimates and percentile-based confidence limits (**Table 12**). The proportion of breeding females on the calving ground (breeding females / (breeding females+non-breeding females+bulls+yearlings)) as well as other cohorts, were estimated. The proportion of adult cows was highest on the photo-north and photo-south strata with lower (<28%) in visual strata. The proportion bulls were relatively high (>50%) in visual strata.

Table 11. Summary of composition data by stratum collected for the June 2022 Qamanirjuaq caribou abundance survey.

Strata	**n**	**Breeders**	**Non-breeders**				**Total caribou**
		Cows^A	**Cows^B**	**Bulls**	**Yearlings**	**Total**	**Breeder & non-breeders**
Photo North	27	3204	270	220	149	639	3843
Photo South	87	3025	706	902	248	1856	4881
Vis1	78	80	187	810	197	1194	1274
Vis2	35	8	16	171	18	205	213
Vis3	33	13	45	120	37	202	215
Vis4	52	4	85	428	101	614	618

[A] As indicated by presence of a calf, antlers, or an udder.

[B] As indicated by absence of calf, an udder or antlers (UC0 in database).

Table 12. Estimates of proportions of various cohorts from composition surveys flown during the June 2022 Qamanirjuaq caribou herd calving-ground abundance survey.

Strata	Estimate	SE	95% CI		CV
Proportion breeding cows (breeding_cows/(breeding_cows+non_breeding cows+bulls+yearlings)					
Photo North	0.904	0.024	0.847	0.940	2.6%
Photo South	0.764	0.038	0.678	0.824	5.0%
Vis1	0.210	0.019	0.172	0.249	9.2%
Vis2	0.113	0.032	0.054	0.181	28.5%
Vis3	0.270	0.040	0.193	0.350	14.7%
Vis4	0.144	0.020	0.106	0.185	14.1%
Proportion adult cows (cows/(cows+bulls+yearlings)					
Photo North	0.834	0.040	0.741	0.895	4.8%
Photo South	0.620	0.055	0.488	0.704	8.9%
Vis1	0.063	0.014	0.039	0.093	22.4%
Vis2	0.038	0.016	0.011	0.071	42.2%
Vis3	0.060	0.022	0.021	0.111	37.0%
Vis4	0.006	0.003	0.001	0.013	48.1%
Proportion of bulls (bulls)/(cows+bulls+yearlings))					
Photo North	0.057	0.015	0.034	0.092	26.2%
Photo South	0.185	0.035	0.132	0.264	18.9%
Vis1	0.636	0.023	0.590	0.684	3.7%
Vis2	0.803	0.039	0.721	0.876	4.9%
Vis3	0.558	0.046	0.473	0.649	8.2%
Vis4	0.693	0.031	0.635	0.758	4.5%

4.5 Estimates

4.5.1 Estimates of total caribou on the calving ground

Estimates of the total number of caribou on the annual core calving ground and peripheral calving strata using both the visual and photo-survey data, are displayed in **Table 13**. Estimates in most strata had coefficients of variation (CV) levels of less than 20% with the exception of the photo-north stratum which had a CV of 21.4% which was due to high variation in densities observed within the strata (**Figure 23**). Vis2 also had a CV of greater than 20%. However, the estimate of abundance was low and therefore it did not contribute significantly to the overall estimate. The resulting precision of the overall estimate of 213,079 caribou on the annual core calving ground was relatively precise with a CV of 11.5%.

Table 1. Estimates of caribou (1+year old) on the annual core calving ground from the core photo (Photo-North & Photo-South), core visual (Vis-1 & 4), and peripheral visual strata (Vis-2 & 3).

Strata	N	SE	95% CI		CV	df
Photo North	98,614	21135.8	62,594	155,360	21.4%	14.0
Photo South	79,193	12212.6	57,456	109,155	15.4%	19.0
Vis1	23,083	1640.9	19,886	26,795	7.1%	18.1
Vis2	2,880	597.9	1,822	4,552	20.8%	10.0
Vis3	2,996	351.2	2,309	3,887	11.7%	10.0
Vis4	6,313	401.1	5,405	7,374	6.4%	6.0
Total	**213,079**	**24478.6**	**166,781**	**272,229**	**11.5%**	**14.4**

4.5.2 Estimates of breeding females and other cohorts on the core breeding ground.

Estimates of the proportion of breeding females (**Table 12**) were then multiplied by the number of caribou on each stratum (**Table 13**) to derive a breeding female estimate of 133,125 (95%-CI=96,561-183,534) (**Table 14**). The estimate of adult cows (breeders and non-breeders) was 156,540 (**Table 15)** (95%-CI=116,635-210,099) suggesting that roughly 23,000 cows on the core calving ground were non-breeding (as determine by lack of calf, antler, or udder). The photo stratum, which was classified as having 90% and 76% adult females (Photo-North and Photo-South respectively), contributed the most to the overall estimate of breeding females and non-breeding females. Relatively few adult females (breeding or non-breeding) were found within the visual abundance strata.

Table 2. Estimates of breeding females from composition data and survey strata estimates (CG= Calving ground).

Strata	Caribou on C.G.		Proportion breeders		Breeding female estimate				
	N	CV	Estimate	CV	N	SE	95% CI		CV
Photo North	98,614	21.4%	0.834	4.8%	82,244	18063.8	51,633	131,004	22.0%
Photo South	79,193	15.4%	0.620	8.9%	49,100	8738.0	33,929	71,054	17.8%
Vis1	23,083	7.1%	0.063	22.4%	1,454	341.1	894	2,364	23.5%
Vis2	2,880	20.8%	0.038	41.7%	109	51.0	40	294	46.8%
Vis3	2,996	11.7%	0.060	37.3%	180	70.3	78	417	39.0%
Vis4	6,313	6.4%	0.006	51.9%	38	19.8	11	126	52.1%
Total	**213,079**	**11.5%**			**133,125**	**20069.4**	**96,561**	**183,534**	**15.1%**

Table 3. Estimates of adult females from composition data and survey strata estimates.

Strata	Caribou on C.G.		Proportion adult females		Adult female estimate				
	N	CV	Estimate	CV	N	SE	95% CI		CV
Photo North	98,614	21.4%	0.904	2.6%	89,147	19251.9	56,395	140,921	21.6%
Photo South	79,193	15.4%	0.764	5.0%	60,503	9803.2	43,197	84,743	16.2%
Vis1	23,083	7.1%	0.210	9.1%	4,847	561.1	3,804	6,176	11.6%
Vis2	2,880	20.8%	0.113	28.4%	325	114.6	152	697	35.3%
Vis3	2,996	11.7%	0.270	14.7%	809	152.0	534	1,225	18.8%
Vis4	6,313	6.4%	0.144	14.1%	909	141.0	623	1,326	15.5%
Total	**213,079**	**11.5%**			**156,540**	**21612.7**	**116,635**	**210,099**	**13.8%**

4.5.3 Extrapolated estimate of total herd size

A composition survey of the Qamanirjuaq fall rutting range was conducted October 2016 to obtain an estimate of the proportion of females in the Qamanirjuaq caribou herd. With no recent fall compositions available, we used the 2016 results to determine the proportion of females within the Qamanirjuaq herd to generate the whole herd estimate. During the fall 2016 composition survey, two main breeding aggregations of caribou were classified with 6,419 and 9,894 bulls and cows classified respectively. The resulting estimates of bull-cow ratios, and overall proportion of cows (cows/(bulls+cows)) are given in **Table 16** (Campbell et al. 2018).

Table 4. Fall 2016 composition survey results.

Ratio	Estimate	SE	Conf. Limit		CV
Bull/cow ratio	0.616	0.026	0.566	0.664	4.1%
Proportion cows	0.619	0.010	0.601	0.639	1.6%
Calf-cow ratio	0.391	0.008	0.376	0.407	2.0%

In 2014, an alternative estimate of herd size was derived by assuming that all adult cow caribou were on the core calving ground (Campbell et al. 2012; Boulanger et al. 2015). This avoided the need of a pregnancy rate since it was assumed that all non-pregnant cows (1.5 years old and older) were on the annual core area, a method that is now successfully integrated into the Qamanirjuaq survey analysis and adopted by other mainland migratory caribou herds across Nunavut and the NWT. Using this method, the estimate of the herd is simply the estimate of females divided by the proportion of females in the herd (**Table 17**). This estimate still pertains to adult caribou and not yearlings (calves of the previous year). The resulting whole herd estimate for the June 2022 Qamanirjuaq calving ground abundance survey is 252,892 (95%-CI=188,050-340,092) 1.5+ year old caribou.

Table 5. Extrapolated population estimates for the Qamanirjuaq herd using estimates of females on the calving ground (CG) and proportion females estimated in fall composition surveys.

Survey data	Estimate	SE	CV	95% Conf. Limit	
Number of caribou on core and peripheral cg	213,079	24478.6	11.5%	166,781	272,229
Number of females (breeding+non-breeding) in core calving ground	156,540	21612.7	13.8%	116,635	210,099
Proportion females in the entire herd	0.619	0.010	1.6%	0.601	0.639
Total estimate of adult (1.5+ yr old caribou) in the herd	252,892	35153.7	13.9%	188,050	340,092

4.5.4 Estimates of trend

Various metrics can be used to estimate trends in ungulate abundance (**Figure 26**). Of these the most robust metric for some herds, including the Qamanirjuaq herd, is adult females. Breeding females will be influenced by yearly variation in pregnancy rates making extrapolation from this metric less reliable in most cases. The number of caribou on the calving ground will be influenced by how extensive the survey was in targeting bulls and yearlings which often may not occur in the vicinity of the annual core calving area. For example, in June 2008, the primary target of survey efforts on the Qamanirjuaq calving ground was adult females and therefore it is likely that bulls and yearlings were counted less than other years, resulting in a lower estimate. Herd size is based on the adult female estimate and trends in herd size will be proportional to adult females since the same assumed bull-cow sex ratio has been used for all herd size estimates. For this reason, we focus on trend estimates of adult females across all surveys.

The estimate of adult females is composed of breeding and non-breeding females. **Figure 27** shows how the proportion of breeding females varied yearly with a relatively low proportion of breeding females in 2014 and 2008. For consistency between surveys, we compared sequential estimates of adult females using a one-tailed t-test to assess if a significant decline had occurred (**Table 18**). Degrees of freedom were estimated for combined estimates for each year using variances and degrees of freedom from each of the sampled stratum (Thompson, 1992). The difference in estimates was significant for the 2008 to 2014 and the 2008 to 2022 comparison (at $\alpha=0.1$) but not significant for other comparisons. The ratio of successive estimates can also be used to estimate gross and annual change (**Table 19**) with yearly change varying from 0.95 to 1.03 between successive surveys. If the ratio of 2008 to 2022 is used to estimate annual change (λ), the resulting estimate is 0.98 (95%-CI=0.96-1.00).

Overall trend was also estimated using weighted log-linear regression of the adult female estimates from 2008 to 2022, which reveals a non-significant decreasing trend with a yearly λ estimate of 0.98 (CI=0.96-1.00) (**Table 20**). This further suggests a slightly longer-term declining trend of 2% per year which is similar to the ratio of the 2008 to 2022 estimate. However, this estimate was not statistically significant. A plot of regression estimates demonstrates the potential of a decreasing trend when the confidence limits of individual estimates are considered (**Figure 28**).

Table 6. Comparison of adult female estimates from successive surveys using t-tests. Also included is a test comparing 2008 and 2022 estimates. P-values are from a one-tailed t-test (H_o $N_2 \geq N_1$, H_a $N_2 < N_1$).

Years compared	$N_{(year\ 1)}$	SE $N_{(year\ 1)}$	Df $_{(year1)}$	$N_{(year\ 2)}$	SE $N_{(year\ 2)}$	Df $_{(year2)}$	t-test	Df t	p-value
2008-14	215,049	17373.9	35	163,066	13296.4	28	-2.38	61.6	0.010
2014-17	163,066	13296.4	28	178,423	13599.8	27	0.81	54.9	0.789
2017-22	178,423	13599.8	27	156,540	21612.7	18	-0.86	31.8	0.199
2008-22	215,049	17373.9	35	156,540	21612.7	18	-2.11	40.2	0.021

Table 7. Estimates of adult females for 2008, 2014, 2017, and 2022. The gross change in estimates (based on the ratio of successive N estimates) and yearly rate of change is also given.

Year	Estimate	Gross change				Yearly change (λ)		
	N	SE	Estimate	Conf. Limit		Estimate	Conf. Limit	
2008	215,049	17,373.9						
2014	163,066	13,296.4	0.76	0.61	0.95	0.95	0.92	0.99
2017	178,423	13,599.8	1.09	0.88	1.36	1.03	0.96	1.11
2022	156,540	21,612.7	0.88	0.64	1.19	0.97	0.91	1.03

Table 20. Regression estimates of trend (2008-2022). The per capita rate of increase(*r*) is estimated as the slope term with the annual finite rate of increase (λ) estimated as the exponent of *r*.

Parameter	Estimate	SE	95% Confidence Limits		Chi-Square	Pr > ChiSq
Intercept	12.246	0.099	12.049	12.436	124.062	0.000
Year (*r*)	-0.021	0.012	-0.043	0.002	-1.804	0.213
λ	0.979		0.958	1.002		

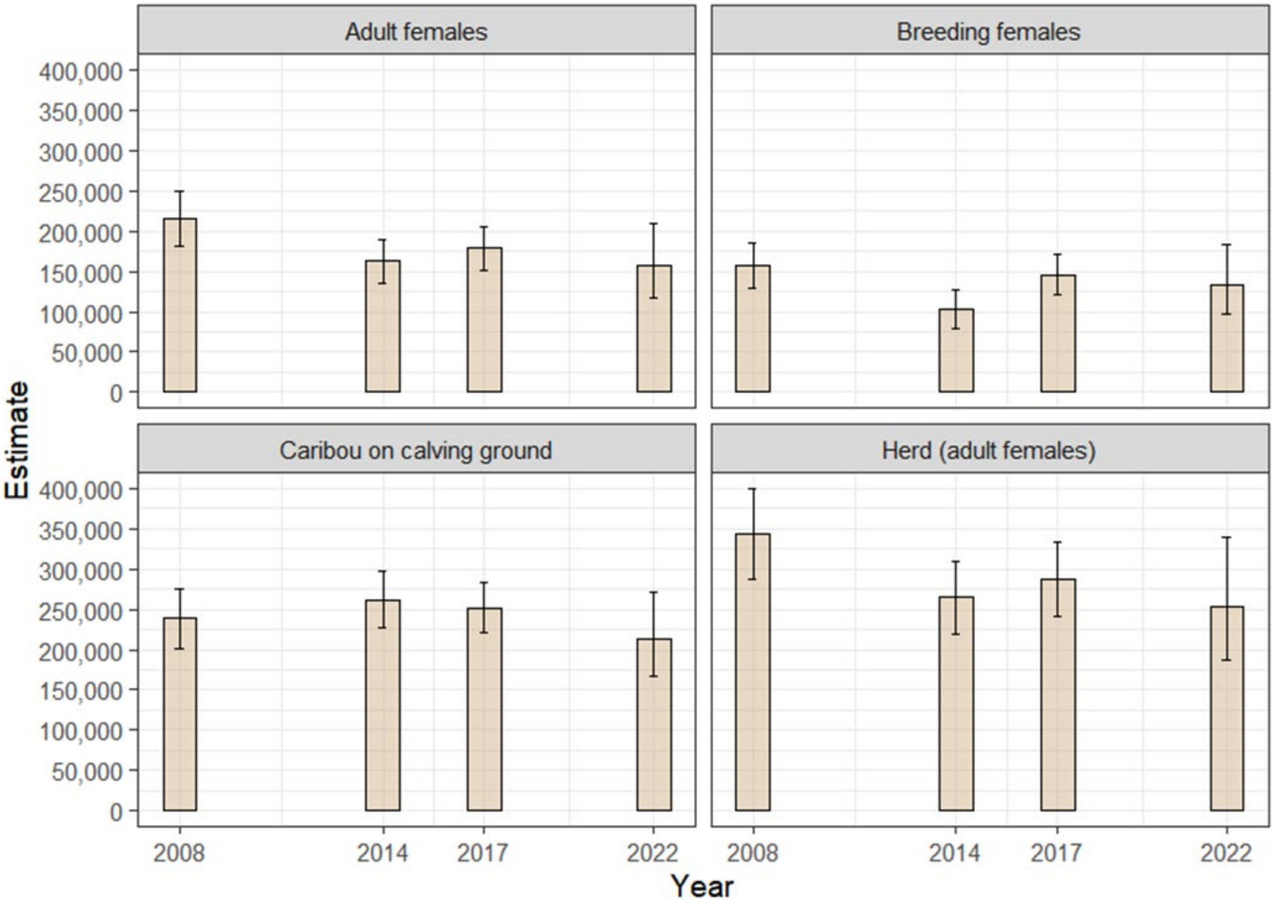

Figure 26. Estimates of Qamanirjuaq herd status using various metrics from 2008 to 2022. Estimates from previous years taken from previous survey reports (Campbell et al. 2010, Campbell et al. 2016, Boulanger et al. 2018).

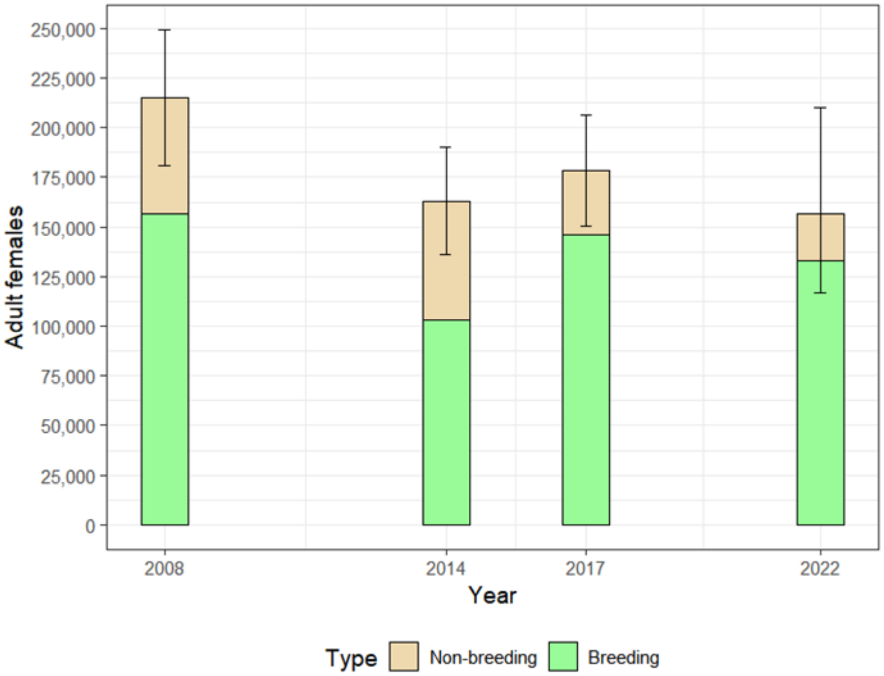

Figure 27. Estimates of Qamanirjuaq adult females, breeding females, and extrapolated herd size based on adult females (**Table 18** for the 2008, 2014,2017, and 2022 surveys).

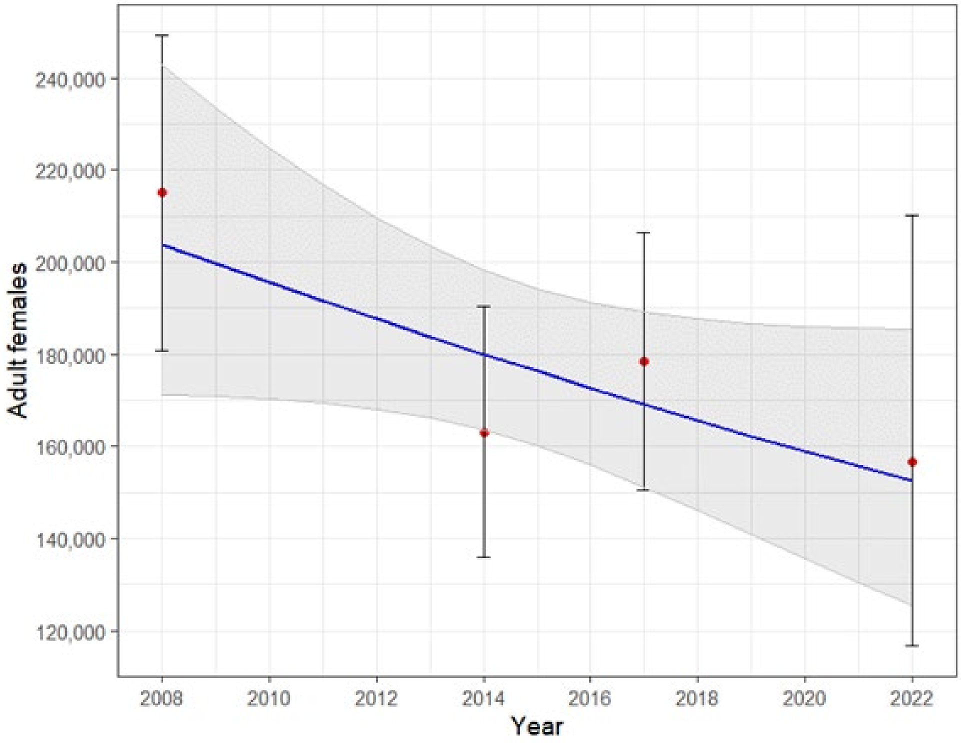

Figure 28. Predicted trends in Qamanirjuaq herd abundance from log-linear regression. Confidence limits on regression predictions are given as hashed blue lines. Individual estimates are shown as red points with displayed 95% confidence limits.

4.6 Changes in Annual Calving Extents

Qamanirjuaq mainland migratory barren-ground caribou typically do not calve in the same locations year after year but rather utilize geographically distinct annual calving grounds within a larger and more geographically predictable concentrated core calving ground (**Figure 29**). Amongst Inuit Qaujimajatuqangit (IQ) and scientific study there is general agreement that the mainland migratory caribou herds, including the Qamanirjuaq herd, cycle every 40 to 60 years (from population low to population low) (Bongelli 2020; Bongelli et al, 2022). Unfortunately, we only have useful telemetry data for the last 30 years, leaving an estimated 20-year gap in our complete understanding of the full extent of the core concentrated seasonal calving range through one complete population cycle (amongst other seasonal ranges). In the case of the Qamanirjuaq herd, both long-term monitoring studies and IQ predict that the spatial extents of annual calving grounds when examined over the long-term will remain within a geographically larger core calving ground, an area representative of the long-term calving needs of the herd (**Figure 29**).

More recently, delineated annual calving grounds have extended beyond the 2013 delineated core calving area. This finding is not entirely unexpected when considering the 20-year gap in our spatial knowledge of Qamanirjuaq herd calving through a full 40-to-60-year population cycle. To understand the herds long-term calving needs in terms of minimum spatial extents, we will need to examine a time series of seasonal range use from population low to population low. To assume we currently hold sufficient information to assess the herds spatial needs for calving, migratory, post-calving, summer, rutting and winter seasonal range, would be misleading and risk the long-term viability of the herd should an underestimate of the long-term core calving requirements of the Qamanirjuaq herd be made.

An examination of telemetry delineated annual calving-grounds for the 2008, 2014, 2017, and 2022 June calving abundance surveys suggest that the annual calving grounds (represented by the 95% utilization distribution) were substantively different, with a general geographic trend to the north and east of 2013 delineated core calving extents when compared to an updated core calving seasonal range polygon current to 2023 using the same analytical methods (**Figure 30**). This same general trend is also seen within a 20-year assessment further broken down into 5-year segments. The results of the 5-year segmented approach suggests that the 4 survey years were representative of more normal environmental patterns and not an artifact of extreme environmental conditions known to have occurred on occasion, within long-term telemetry data (**Figure 31**). We suspect annual core calving range will see further changes as we expand the telemetry database. As more telemetry data is collected, we strongly suspect that annual core calving extents will expand to accommodate annual variations repeated throughout the population cycle. The resulting updated polygons will better predict long-term annual requirements for the herd.

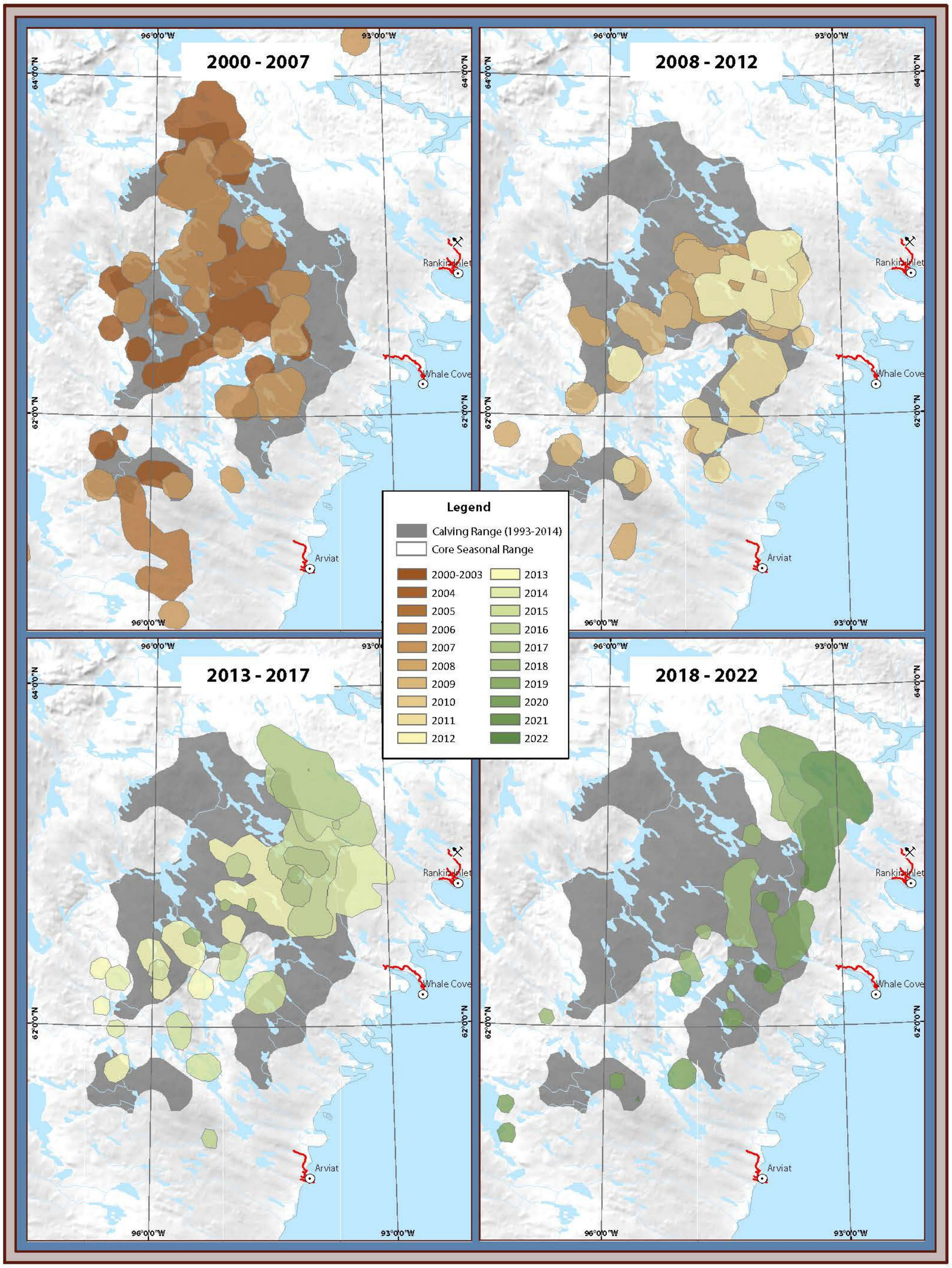

Figure 29. A representation of annual calving grounds from June 9th through June 22nd for the years 2000 through 2022. Polygons developed using a kernel density estimation (KDE) of caribou cow telemetry data.

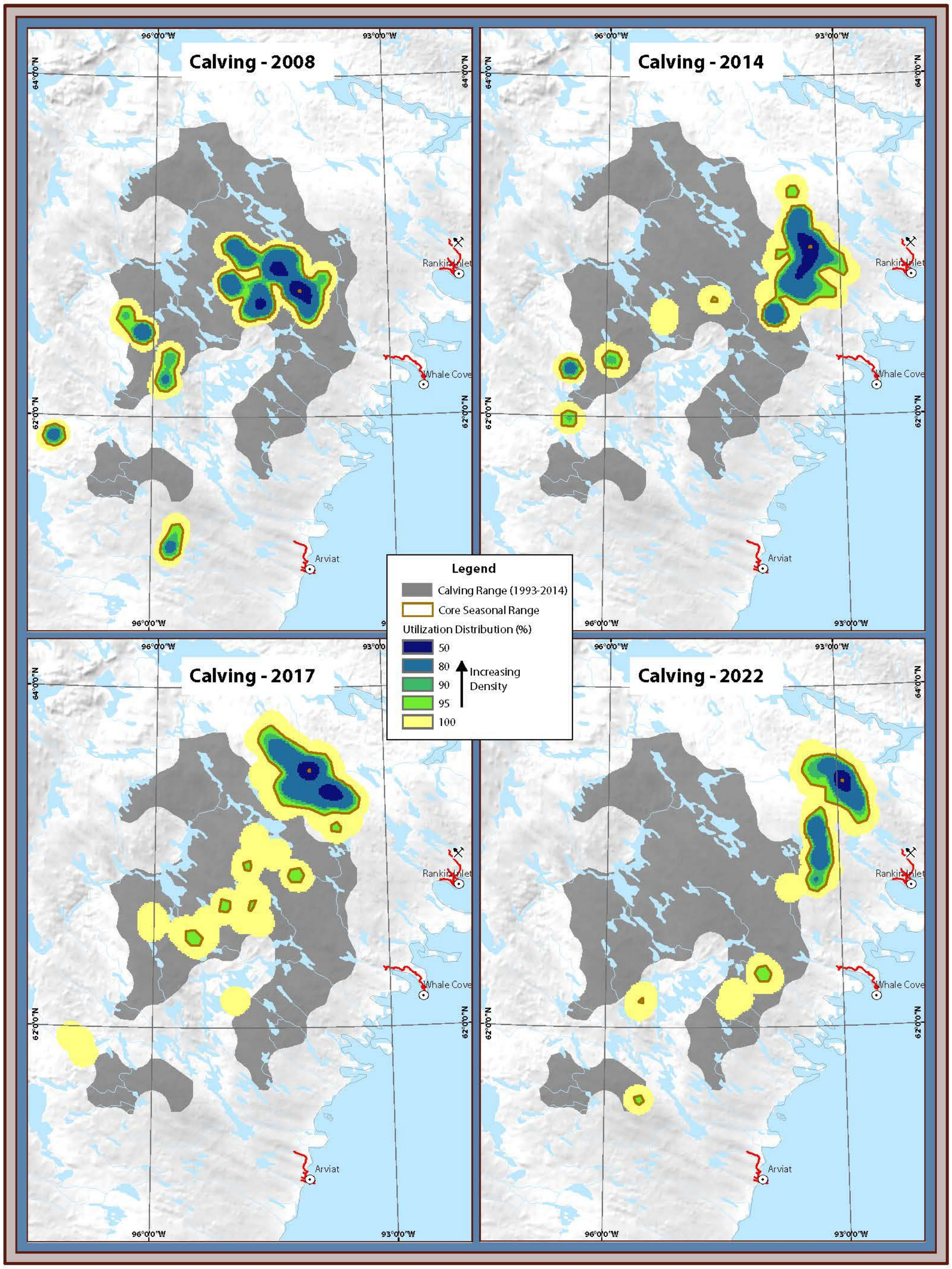

Figure 30. A spatial assessment of Qamanirjuaq Herd annual calving extents for each of the 2008, 2014, 2017, and 2022 (this report) calving-ground abundance surveys. A kernel density estimation (KDE) approach was used to generate utilization distributions.

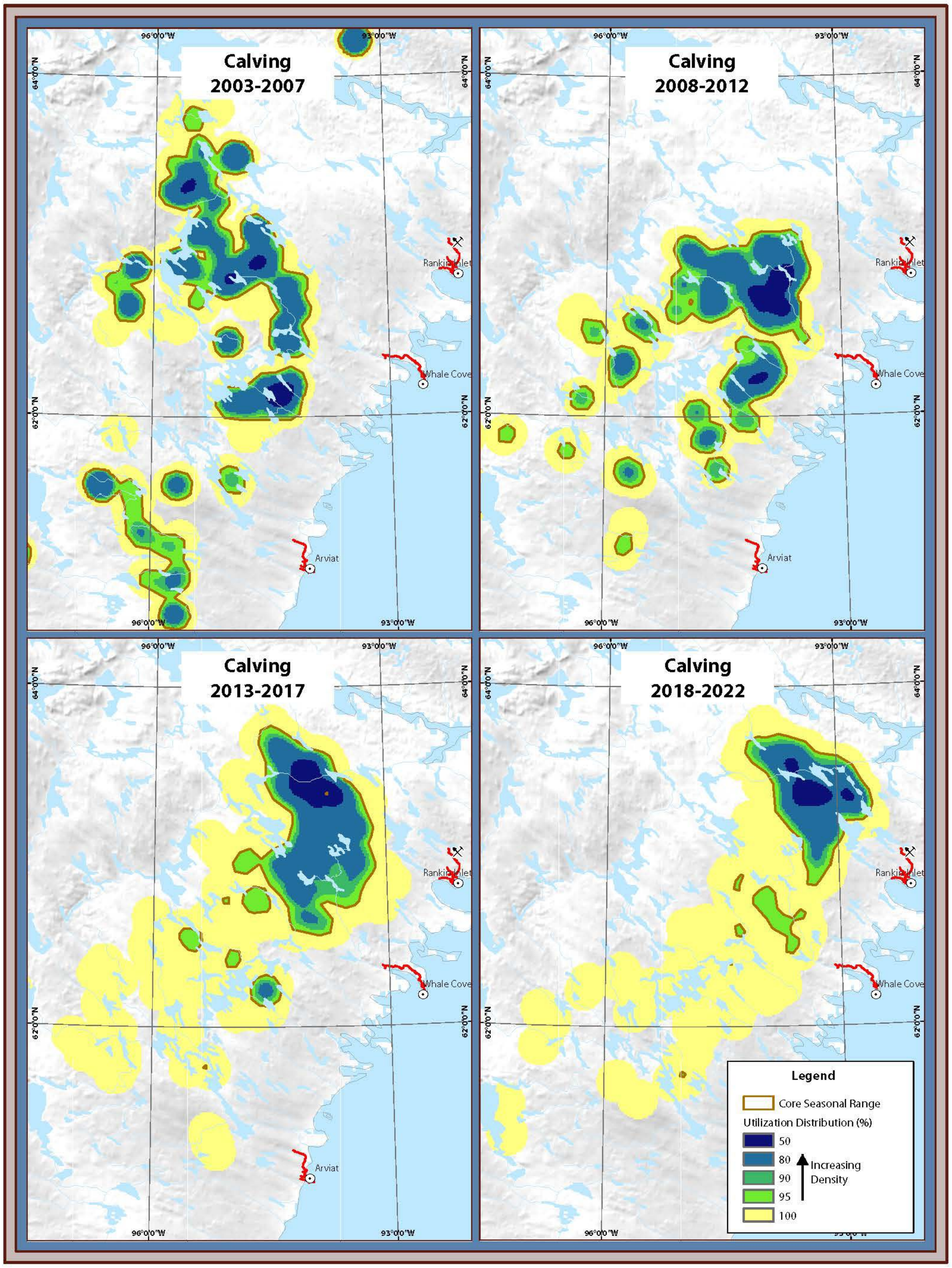

Figure 31. A spatial assessment of Qamanirjuaq Herd annual calving extents over a 20-year period broken down by 5-year increments. A kernel density estimation (KDE) approach was used to generate utilization distributions.

5.0 DISCUSSION

The overall survey results suggest that the herd is relatively stable when compared to the 2014 and 2017 estimates. However, direct comparison with the 2008 estimate along with the 2014 and 2017 estimates, suggests a slow decline. A one-tailed t-test comparison of 2008 and 2022 estimates reveals a significant decline. However, the overall trend from a regression analysis was not statistically significant.

Survey precision was lower in 2022 (CV=14% for adult females) than previous years but still within the acceptable range (CV<20%). The main reason for this lower precision of the photo-north stratum was due to a high level of aggregation (**Figure 23**) which caused high variation in densities amongst individual transects. One approach to address this issue would be to increase the number of lines and subsequent coverage in the stratum. The coverage in the photo-north stratum was reasonably high (42%) when compared with previous surveys. For example, the highest coverage in the 2017 survey was 35% in two of the photo-stratum. The approach of increasing coverage and aggressively sampling likely areas of aggregation with visual strata buffers could be considered as an approach to confront aggregation in future surveys.

Coverage in other caribou surveys, such as the Bathurst in 2012 has been as high as 72% which was due to a highly aggregated group of caribou in a small (914.2 km^2) survey area (Boulanger et al. 2014b). The resulting CV for the estimate in this stratum was 8%. A randomization approach was used to assess if there was an optimal coverage level where the CV did not change with increasing coverage with no asymptote found. A coverage level of 65% was still required to obtain a CV of less than 10%.

Recently, new approaches have been developed that consider multiple sources of information to improve our estimates and understanding of trends. The demographic status of a herd can be better understood using an Integrated Population model (Schaub and Kery, 2022) that uses data from composition surveys, collar survival rates, and calving ground surveys to provide an overall estimate of trend as well as refined estimates of demographic parameters. This approach has been used successfully to provide refined estimates of trend for the Bluenose East and Bathurst herds (Adamczewski et al. 2022, Boulanger et al. 2022). This approach will be pursued for the Qamanirjuaq herd in the future.

Our collective experience from the Bathurst Herd shows that major declines in mainland migratory barren-ground caribou subpopulations must be detected as early as possible to inform decisions to maintain the subsistence needs of Inuit and other indigenous peoples. Knowing the trend and status of the population will provide co-management partners with more options such as critical habitat protection, non-quota limitations (NQLs), commercial harvesting restrictions, in order to maintain herds or promote recovery following any major declines. All monitored indices of the herd, when examined in series, indicate that the Qamanirjuaq herd is in a continued slow decline. Therefore, lack of appropriate actions may exacerbate or prolong or halt herd recovery and place undue hardship on Inuit that harvest this herd both commercially and for subsistence. These hardships will also impact harvesters from Northern Manitoba, Northern Manitoba, and the NWT.

It is well known within the scientific literature, and amongst Northern Indigenous peoples that Barren-ground caribou undergo regular and long-term (between 50 and 70 years) cycles of abundance (Bongelli et al. 2020; Gunn, 2003; Zalatan et al. 2006; St. John, 2022). Additionally, there appears to be synchrony between the barren-ground herds including the Bluenose, Bathurst, Beverly, Leaf River, and George River mainland migratory herds that could be in response to large-scale environmental factors such as climate change and associated localized weather events and patterns, density dependant reproductive disease and parasites, and predator and human harvest effects. This also suggests that these mainland migratory caribou herd declines and associated mechanisms would impact all herds eventually. These impacts could be exacerbated by other anthropogenic effects on caribou mortality such as a growing internet market for caribou meat within Nunavut Territory, or industrial disturbance and associated avoidance behaviour and habitat loss within sensitive seasonal range such as the core calving grounds, post-calving areas and migratory routes (Panzacchi et al. 2012, Plante et al. 2020, Boulanger et al. 2021, Boulanger et al. 2023, Severson et al. 2023). Loss of seasonal range access and availability would likely result in quantitative changes to long-term abundance through the loss of productive annual range and herd fragmentation. The impacts of anthropogenic activities such as industrial development or extractive practices on caribou and their range are uncertain, however, the scientific literature warns of the very real risk of negatively impacting caribou and their range over the long-term. With the industrial extraction of natural resources on the increase in northern North America, and more specifically within barren-ground caribou critical seasonal range, it is becoming more and more important that land use regulators and wildlife managers and biologists develop a better understanding of important seasonal range while closely monitoring herd status in order to more effectively manage anthropogenic impacts on range size and use, migratory behaviour, and herd productivity. Not doing so would work against Article 5, Part 1, Section 5, of the Nunavut Agreement as it could reduce harvesting opportunities, negatively impacting Inuit's right to harvest wildlife to meet their subsistence needs as described in the Nunavut agreement (**Figure 32**).

5.1.5 The principles of conservation are:

- *(a) The maintenance of the natural balance of ecological systems within the Nunavut Settlement Area;*
- *(b) The protection of wildlife habitat;*
- *(c) The maintenance of vital, healthy, wildlife populations capable of sustaining harvesting needs as defined in this Article; and*
- *(d) The restoration and revitalization of depleted populations of wildlife and wildlife habitat.*

Figure 32. Article 5.1.4 of the Nunavut Agreement "*The Principles of Conservation*", an excerpt from the Nunavut Agreement Page 27 (Agreement Between the

Inuit of the Nunavut Settlement Area and Her Majesty the Queen in right of Canada, 2009).

as well as negatively impacting the harvesting rights and cultural practices of Indigenous cultures within Manitoba, Saskatchewan, and the Northwest Territories.

6.0 MANAGEMENT CONSIDERATIONS

Concerns regarding the Qamanirjuaq herd have changed little since the June 2017 Qamanirjuaq caribou calving ground abundance survey. Hunters and Trappers Organizations (HTO's) and the Kivalliq Wildlife Board (KWB) continue to communicate their concerns over the status of the Qamanirjuaq herd. Some of the most common concerns brought forward during annual consultative meetings include: 1) inter-territorial caribou meat sales, primarily between the Kivalliq and Baffin Regions, are believed to be excessive and negatively impacting the local harvest of caribou for community based food needs; 2) there is extreme concern over development in calving grounds, Key Access Corridors (mutually inclusive calving, post-calving, and spring migratory seasonal range), water crossings, and post-calving grounds, as well as concern over disturbance to migrating caribou along linier industrial infrastructure. Both peer-reviewed science and IQ agree that industrial development in calving and post-calving grounds have not been shown to be mitigated in other contexts and areas resulting in changes in caribou abundance, distribution, behaviour, and health over the long term. These impacts, should they occur, would negatively impact caribou and Inuit harvesting; 3) Many hunters from across the region have communicated their sense of a general decline in abundance, and increase in disease prevalence. These concerns suggest that conditions are changing on the Qamanirjuaq range, and that our ability to monitor these changes should be heightened so that all co-managers can effectively advocate effective management action to safeguard Nunavut's largest caribou population.

Though the regression-based trend of Qamanirjuaq herd abundance estimates between the June 2008 and June 2022 indicates a non-significant declining trend of 2% annually (Figure 28 and Table 20), the current survey results show a significant decline in mean herd abundance between June 2008 and June 2022 (Table 18). The lack of statistical significance in regression-based trend estimates (that include 2008, 2014, 2017, and

2022 estimates) may be due to stability in the period of 2014-2017, however, it should be noted that precision of estimates (large confidence intervals) is low along with lower sample sizes of survey points. Spring composition studies when examined together suggest productivity levels consistent with a slow decline. The fact that mainland migratory barren-ground caribou are well known within both scientific (Bongelli et al 2020) and local knowledge to follow long-term (40 to 60 year) population fluctuations suggests that the Qamanirjuaq herd is at high risk of further decline over the next 5 to 10 years as food resources become over utilized and as a result, harder to find.

Because of the uncertainty between the 2014, 2017, and 2022 abundance estimates, this report is not recommending any management action at this time. This report does however acknowledge the slow decline indicated by comparison of June 2008 and 2022 estimates and for this reason highlights the importance of continued monitoring of herd trend. Of equal importance is the protection of critical range to ensure healthy seasonal range remains accessible to Qamanirjuaq caribou when recovering from cyclical and/or significant declines in abundance, typical of mainland migratory barren-ground caribou herds such as the Qamanirjuaq herd. Such protections will act to secure Inuit harvesting rights by respecting the principles of conservation guaranteed to Inuit within the Nunavut Agreement which in turn will maximize harvesting opportunities and the associated health and monetary benefits that result from healthy abundant caribou populations. These actions align with Regional Wildlife Organization (RWO), HTO, and community priorities to protect caribou annual core calving areas, key access corridors, post-calving range, water crossings, and other important seasonal range across the Qamanirjuaq herd's annual range.

Initial survey results and progress have been shared with co-management partners, including representatives from the KWB, Kivalliq HTO's, Beverly Qamanirjuaq Caribou Management Board (BQCMB), Nunavut Wildlife Management Board (NWMB), and the jurisdictions of Saskatchewan, Manitoba, NWT and Canada (ECCC). Consultations with community Hunters and Trappers Organizations regarding survey results occurred in early 2024, however, co-management partners including HTO's, the Kivalliq Wildlife Board (KWB), the Nunavut Wildlife Management Board (NWMB), and the Nunavut Department of Environment (ENV), will continue to discuss survey results and recommendations (as represented in this report) and other issues at upcoming annual meetings. The GN ENV will continue to include, investigate, and report on, any findings derived from new scientific analysis, and IQ that may provide additional insight into the mechanisms influencing Qamanirjuaq caribou herd abundance and trend.

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ᑎᑎᖅᑐᒐᖅ 1. ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑦ (*Reinhardtius hippoglossoides*).

ᐅᖃᐅᓯᖃᒃᓯᒪᓂᖏᑦ

ᐃᖃᓗᓕᕆᓂᐅᔪᖅ ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᓐᓂᑦ (GHL) ᐱᑕᖃᖅᑐᖅ ᓯᓚᑖᓂ ᓄᓇᕗᒻᒥ ᐊᖏᖅᑲᑎᒌᒍᑎᑖᒍᑦ ᓄᓇᖁᑎᒋᔭᐅᔪᓄᑦ (NSA) ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ (NAFO) ᐊᕝᒍᑐᖅᓯᒪᔪᖅ (SA) 0 ᐊᕝᒍᑐᖅᓯᒪᔪᖅ ᐅᐊᓐᓇᒧᑦ, ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0A (ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓ) ᐊᒻᒪ ᓂᒋᖓ, ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0B (ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓᓐᓂ). ᑮᓇᐅᔭᓕᐅᕿᓯᐊᓂᕐᒧᑦ ᐃᖃᓗᒐᓱᐊᕐᓇᐅᔪᖅ ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᓐᓄᑦ ᐱᒋᐊᕐᓂᖅᑐᑦ ᔮᓄᐊᕆ 1 ᐊᒻᒪ ᐃᓱᓕᓐᓂᖅᑐᑎᑦ ᑎᓯᐱᕆ 31. ᓄᓇᙳᐊᖅ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᖅ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᐊᕝᒍᑐᖅᓯᒪᔪᑦ ᐊᒻᒪᓗ ᐊᕕᑦᑐᖅᓯᒪᔪᑦ ᐊᑑᑎᔪᑦ ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᓐᓂᑦ ᐃᖃᓗᓕᕆᓂᕐᒧ ᓇᓂᓴᐅᔪᓐᓇᖅᑐᒃ ᐃᓚᒍᑕᖓ 1–ᒥ.

ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑦ ᐃᖃᓗᐃᑦ ᐅᕙᓂ ᐊᕝᒍᑐᖅᓯᒪᔪᖅ 0 ᐃᓚᒋᔭᖓ ᑭᓪᓕᖏᓐᓂᑦ ᐅᖓᑕᐅᔾᔨᖃᑦᑕᐅᑎᔪᑦ ᐃᖃᓗᐃᑦ ᑲᑐᑎᔭᐅᔪᑦ ᐅᑯᓂᖓ ᑲᓇᑕ (ᐊᕕᑦᑐᖅᓯᒪᔪᑦ 0A ᐊᒻᒪ 0B) ᐊᒻᒪ ᐊᑯᑭᑐᑦ (ᐊᕕᑦᑐᖅᓯᒪᔪᑦ 1A ᑎᑭᑦᑐᒍ F ᐃᒪᐃᓐᓇᕐᒥᑦ). ᑐᔅᓯᕚᖑᔪᓂᑦ ᑕᒪᒃᑭᓐᓄᒃ ᓄᓇᓕᕐᔪᐊᓐᓄᒃ, ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ (SC) ᐱᑕᖃᖅᑎᑦᑎᔪᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᒻᒥᑦ ᐃᖃᓗᖁᑕᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ ᐊᒻᒪᓗ ᒪᑭᒪᑎᑕᐅᓂᖏᓐᓄᑦ ᐃᖃᓗᑦᑕᐅᕙᑦᑐᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ. ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᐅᕐᓂᖃᖏᑦᑐᑦ ᑖᒃᑯᓄᖓ ᐃᖃᓗᖁᑕᐅᔪᓄᑦ; ᑲᓇᑕ ᐊᒻᒪ ᐊᑯᑭᑦᑐᑦ ᑲᒪᒋᔭᖃᖅᑐᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᐅᓂᕐᒥᑦ ᐃᒪᖏᓐᓂᑦ ᐊᑐᓂ ᐃᒪᖁᑐᖏᓐᓂ. ᑲᓇᑕ ᐊᒻᒪᓗ ᐊᑯᑭᑐᑦ ᐊᑯᓂᐅᔪᒥᒃ ᐊᖏᖅᑲᑎᒌᒃᓯᒪᔪᑦ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAC) ᖁᕝᕙᓯᓐᓂᖏᑦ ᓴᖅᑭᖅᑎᑕᐅᓯᒪᓗᑎᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᐅᖃᖅᓯᒪᔪᑦ ᐊᕕᑦᑕᐅᓗᑎᑦ 50/50 ᑖᒃᑯᓄᖓ ᒪᕐᕌᖕᓄᒃ ᓄᓇᓕᕐᔪᐊᓐᓄᕐ.

ᐃᖃᓗᒐᓱᖕᓂᖅ 2024–ᒥ, ᐃᓂᒋᔭᐅᔪᖅ ᐊᕝᒍᑐᖅᓯᒪᔪᖅ 0 ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑦ ᑲᑎᔅᓱᒋᑦ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐋᖅᑭᒃᑕᐅᓚᐅᖅᑐᖅ 16,502.5 ᑕᓐᔅ (t) ᐅᕿᖏᓐᓂᕐᒧ. ᑲᑎᖅᓱᖅᓯᒪᔪᖅ ᒫᓐᓇᐅᔪᖅ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐊᕕᑦᑐᖅᓯᒪᔪᓂᒃ ᐊᑯᓐᓂᖏᓐᓂ ᑕᑯᑎᑕᐅᔪᖅ ᓇᓗᓇᐃᒃᑯᑎ 1–ᒥ.

ᓇᓗᓇᐃᒃᑯᑎ 1. ᑲᑎᖅᓱᖅᓯᒪᔪᖅ ᒫᓐᓇᐅᔪᒥ ᑲᑎᑦᑐᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᓐᔅ (t) ᐅᕿᖏᓐᓂᓚ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐊᕕᑦᑐᖅᓯᒪᔪᓂᒃ ᐊᑯᓐᓂᖓᓂ.

ᐃᖃᓗᒐᓱᐊᕐᕕᒋᔭᐅᔪᑦ	ᐅᒥᐊᕐᔪᐊᑦ/ᐱᔪᒪᓂᖃᖅᑐᑦ	2024 ᐃᓕᐅᖅᑲᖅᑕᐅᔪᑦ (ᑕᓐᔅ)
ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0A	ᓄᓇᕗᑦ ᐊᔾᔨᐅᖏᑦᑐᓄ ᑐᓂᔭᐅᓯᒪᔪᑦ	8,604.99
	ᓄᓇᖅᐸᓯᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᕚᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ	100.00
	ᑲᑎᑦᑐᒋᑦ ᐃᓂᐅᔪᒥ 0A ᑰᑕᐃ	**8,704.99**
ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0B	ᓄᓇᕗᑦ ᐊᔾᔨᐅᖏᑦᑐᓄ ᑐᓂᔭᐅᓯᒪᔪᑦ	3,840.59
	ᓄᓇᕕᒃ ᐊᔾᔨᐅᖏᑦᑐᓄ ᑐᓂᔭᐅᓯᒪᔪᑦ	402.82
	ᐱᓕᕆᕕᐅᔪᑦ ᐊᒻᒪᓗ ᐊᔾᔨᐅᖏᑦᑐᓄ ᑐᓂᔭᐅᓯᒪᔪᑦ ᑎᒍᒥᐊᖅᑏᑦ	2,654.09
	ᐋᖅᑭᔅᓯᒪᔪᓄᑦ ᐃᖃᓗᒐᓱᐊᕈᑏᑦ ᐊᑭᑦᑐᕋᐅᑎᓂᖅ	900.00
	ᑲᑎᑦᑐᒋᑦ ᐃᓂᐅᔪᒥ 0B ᑰᑕᐃ	**7,797.50**

2023–ᒥ, ᐃᓂᐅᔪᒥ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0 ᑲᑎᑦᑐᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᒪᐃᓐᓂ ᐃᖃᓗᒃᑕᐅᓚᐅᖅᑐᑦ.

ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0A ᐊᑯᑭᑦᑐᓂ ᖃᓕᕋᑖᑦ ᐃᖃᓗᓕᕆᓂᖅ ᓂᐅᕐᕈᑕᐅᓲᑦ ᐃᖃᓗᓕᕆᓂᖅ ᑐᕌᖓᓪᓚᕆᒃᑐᖅ ᓄᓇᕗᒻᒧᑦ (NU) ᐱᔪᒥᒋᔭᐅᔪᑦ. 2006–ᒥᑦ ᐊᒻᒪ ᒪᓕᓐᓂᖃᖅᑐᑎᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐱᕚᓪᓕᐊᓂᖓ ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᒻᒥᑦ (NA), ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ (ᑲᑎᒪᔩᑦ) ᐱᖁᔨᕗᖔᓕᐅᓚᐅᖅᑐᑦ ᐃᓕᐅᖅᑲᖅᑕᐅᓗᑎ 100 t ᐊᕕᑦᑐᖅᓯᒪᔪᒧᑦ 0A ᐃᒪᕕᐅᑉ ᑰᑕᖏᑦ ᓄᓇᖅᐸᓯᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᕚᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ ᓄᓇᕗᒻᒥ ᓇᔪᖅᑕᖓᓂ; ᐱᖁᔨᕗᖔᕈᑦ ᐃᑲᔪᓱᖅᑕᐅᓚᐅᖅᑐᖅ ᒥᓂᔅᑕᒧᑦ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒥ (ᒥᓂᔅᑕ) ᐊᒻᒪ ᐃᓕᐅᖅᑲᖅᑕᐅᓚᐅᖅᑐᑦ 2006 ᐃᖃᓗᒐᓱᐊᕐᓇᐅᑎᓪᓗᒍ. ᑕᐃᑲᙵᓂᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᙵᓚᐅᖅᑎᓪᓗᒍ, 100 t ᓄᓇᖅᐸᓯᒻᒥ ᑰᑕᐅᔪᑦ ᒪᓕᒃᓯᒪᔪᑦ ᑲᒪᒋᔭᐅᖏᓐᓇᖅᓱᑎᒃ ᐊᒻᒪᓗ ᐲᔭᖅᑕᐅᕙᒃᓱᑎᒡᓗ ᐊᕕᑦᑐᖅᓯᒪᔪᓂ 0A ᐃᒪᕕᐅᑉ ᑰᑕᖏᓐᓂ. ᑲᑎᒪᔩᑦ ᑎᑎᖅᑲᖓᓂ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒥ (DFO) ᒥᓂᔅᑕ ᓛᐳᑎᓕᔭᐃᒧ ᓯᑎᐱᓇ 14 2023–ᒥ, ᑲᑎᒪᔩᑦ ᑎᑎᕋᓚᐅᖅᑐᑦ ᐱᒻᒪᕆᐅᓂᖓᓂ 100 t ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0A ᓄᓇᖅᐸᓯᒻᒥ ᑐᓂᔭᐅᓯᒪᔪᑦ ᐊᑐᖅᑕᐅᓂᖓᑕ ᕿᒥᕐᕈᔭᐅᓂᖓᓂ ᐊᒻᒪᓗ ᖃᐅᔨᓴᖅᑕᐅᓂᖓᓂ. ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒥ ᑲᑎᒪᔩᑦ ᓴᓇᔨᖏᓂ ᐅᖃᓪᓚᖅᑎᖅᑕᓯᓂᕐᒥ ᐱᒋᐊᖅᓯᒪᔪᖅ ᑖᔅᓯᒪ ᒥᒃᓵᓄᑦ ᐊᒻᒪᓗ ᐊᖏᖅᓯᒪᓂᖅᑲᖅᑐᓂ ᓴᓇᖅᑲᑎᖅᑲᖅᓂᕐᒥ ᓄᓇᖅᐸᓯᒻᒥ ᐃᖃᓗᓕᕆᓂᖅ ᐱᕚᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ ᐱᕕᒃᓴᐅᔪᑦ ᐊᑲᐅᓂᖅᐹᖑᖁᓪᓗᒋ.

ᑭᓪᓕᓯᓂᐊᖅᑕᐅᓯᒪᔪᑦ ᑐᓴᐅᒪᔾᔪᑏᑦ

ᕖᕝᕗᐊᕆ 2024–ᒥ, ᑲᓇᑕ ᐊᒻᒪᓗ ᑕᓐᒪᐅᒃ (ᐱᓕᕆᔾᔪᔾᔨᓪᓗᑎᒃ ᐊᑯᑭᑦᑐᑦ) ᑐᒃᓯᕋᖅᓯᒪᔪᑦ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᑦ (SC) ᐊᑐᐃᓐᓇᕈᐃᓗᑎᒃ ᐊᑕᖐᓯᒪᔪᒥᒃ ᖃᐅᔨᓴᕐᓂᕐᒥ ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑦ ᐃᖃᓗᖁᑎᖏᑕ ᖃᓄᐃᑕᖓᓂᖏᓐᓄᑦ ᐊᒻᒪᓗ ᐃᓚᒃᑯᓯᐅᔪᑦ ᑲᑎᑦᑐᒋ ᐱᑕᖃᕐᕕᐅᔪᑦ ᑕᐅᕙᓂᑐᒃᑖᖅ ᐱᑕᖃᕐᕕᐊᓂ ᐊᒻᒪᓗ ᐅᖅᑲᐅᔨᕝᕕᐊᖏᓐᒥᑦ ᑲᑎᑦᑐᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖁᕙᓯᕐᓂᖏᖓᓂ ᑐᕌᖓᔪᓂ 2025–ᒧ ᐊᒻᒪᓗ 2026–ᒧ. ᓯᓚᒃᑲᓐᓂᖅ, ᑐᒃᓯᕋᖅᑐᖅᑲᓚᐅᖅᑐᑦ ᐃᖃᓗᒐᓱᐊᑦ ᖃᐅᔨᓴᖅᑐᖅᑲᖅᑐᑦ ᐃᓱᒪᓕᐅᕐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᔾᔪᑕᐅᔭᓇᐊᓚᑦ ᐊᑯᓂᐅᔭᒃᑯᑦ ᑲᔪᓯᑎᑦᑎᓇᓱᓪᓗᒋᑦ ᐊᒻᒪᓗ ᐅᖅᑲᐅᔨᕝᕕᐊᐱᐊ ᑐᓴᖅᑎᑦᑎᔾᔪᑕᐅᔪᑦ ᐊᓯᑦᑐᕐᓂᖅᑕᐅᓂᕐᒧᑦ ᑖᒃᑯᐊ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐅᐃᓯᕐᓱᐃᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ (PA) ᐊᐅᓚᑦᑎᓂᖅᑲᕐᓂᖏᑦ.

ᓯᕗᓪᓕᖅᐸᓯ, ᐃᓄᒋᐊᖕᓂᐅᔪᓄᑦ ᐱᓕᕆᔾᔪᑎᒧᑦ (Stochastic Production in Continuous Time [ᑕᐃᒪᓐᓇᒥᐊᖅᑐᖅ ᓴᖅᑭᑎᑦᑎᔨ ᓯᕿᙳᔭᒥ ᑲᔪᓯᔪᒥ], SPiCT ᐱᓕᕆᔾᔪᑎ) ᐊᖏᖅᑕᐅᓚᐅᖅᑐᖅ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᐃᓂᐅᔪᒧ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0+1 (ᐃᒪᕕᒃ) ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑦ ᐃᖃᓗᖁᑕᐅᔪᓄᑦ ᐊᒻᒪᓗ ᐊᑐᖅᑕᐅᓪᓗᓂ ᖃᐅᔨᓴᕐᓂᕐᒥ ᑖᒃᑯᐊ ᐃᖃᓗᖁᑕᐅᔪᑦ ᖃᓄᐃᑕᖓᓂᖏᖓᓂ ᔫᓂ 2024 ᑕᖅᑭᖓᓂ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᑲᑎᒪᑎᓪᓗᒋ. ᐱᕕᐅᔪᑦ ᑕᒪᑐᒧᖓ ᐱᓕᕆᔾᔪᑎᒧᑦ ᐱᖃᓯᐅᔾᔨᔪᑦ ᓄᓈᕐᓂᕐᒥ ᑎᑎᖅᑲᖁᑏᑦ ᐊᒻᒪᓗ ᐊᑐᖅᑕᐅᕙᒃᑐᑦ ᓇᓗᓇᐃᔭᐅᑏᑦ ᑖᒃᑯᓇᙵᑦ ᐱᕕᐅᔪᐊᓐᓇᖅᑐᓂᒃ ᐃᖃᓗᖁᑕᐅᔪᑦ ᐆᒪᔪᑦ ᐅᕿᖏᓐᓂᓚ ᐊᕐᒍᒍᑕᒫᑦ ᕿᒥᕐᕈᔭᐅᓂᕐᒧ ᑎᑎᖅᑲᖁᑎᓂ.

ᐃᖅᑲᓗᖁᑕᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄ ᕿᒥᕐᕈᔭᐅᔪᑦ ᐱᓕᕆᔭᐅᓚᐅᖏᑦᑐᑦ 2018–ᒥ, 2020–ᒥ, ᐅᕝᕙᓘᓐᓃᑦ 2021–ᒥ ᓲᖃᐃᒻᒪ ᕿᒥᕐᕈᔾᔪᑎᒧ ᐅᒥᐊᕐᔪᐊᖅ ᐊᑐᖅᑕᐅᓚᐅᖏᒻᒪ 2018 ᐅᑭᐅᖓᓂ ᐊᒻᒪᓗ ᐅᒥᐊᕐᔪᐊᖅ ᐃᓇᖏᖅᓯᓂᐊᖅᑐᖅ ᐱᔭᕇᖅᑕᐅᓚᐅᖅᖢᓂ 2022 ᐅᐱᕐᖔᒥ. ᕿᒥᕐᕈᓂᖅ ᐱᔭᕇᖅᑕᐅᓚᐅᖅᑐᖅ 2019–ᒥ ᓂᐅᕐᕈᑎᖃᖅᐸᒃᑐᓄᑦ ᐅᒥᐊᕐᔪᐊᒥ ᐊᑐᖅᖢᑎᒃ, ᑭᓯᐊᓂ ᐃᓱᒪᒋᔭᐅᓚᐅᖅᑐᖅ ᐊᔾᔨᖃᖏᓐᓂᖓᓂ ᑭᖑᓂᑦᑎᓐᓂ ᕿᒥᕐᕈᓂᐅᔪᕕᓂᕐᓂ. ᐱᓕᕆᖃᑎᒌᖕᓂᖅ ᐱᑕᖃᙱᑎᑦᑐᒍ ᑭᖑᓂᑦᑎᓐᓂ ᕿᒥᕐᕈᓂᕐᒧ ᐅᒥᐊᕐᔪᐊᖅ (*RV Paamiut*) ᐊᒻᒪᓗ ᓄᑖᖅ ᕿᒥᕐᕈᓂᕐᒧ ᐅᒥᐊᕐᔪᐊᖅ (*RV Tarajoq*) ᐊᑯᓐᓂᖓᓂ ᑐᑭᖃᖅᑲᑐᖅ ᑎᑎᖅᑲᖅᑯᑎᓂ ᑕᒪᐃᓐᓂ ᑖᒃᑯᓇᙵ ᒪᕐᕉᖕᓂ ᐅᒥᐊᕐᔪᐊᓂ ᐊᑐᖅᑕᐅᔪᓐᓇᖏᑦᑐᑦ ᕿᒥᕐᕈᔭᐅᖁᑦᑐᒋ ᐊᑯᓂᐅᔪᒃᑯᑦ ᐱᓕᕆᓂᐅᔪᓂ ᐃᖅᑲᓗᖁᑕᐅᔪᓄᑦ ᐊᐅᓚᓂᓇᕙᒃᑕᖏᑦᑕ ᒥᒃᓵᓄᑦ, ᓇᓗᓇᕐᓂᐅᔪᒥ ᐊᓯᐃᓛᒃ ᓴᖅᑭᑎᑦᑎᑦᑐᓂ ᐃᖅᑲᓗᖁᑕᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦᑕ ᒥᒃᓵᓄᑦ. ᑭᒡᓕᓯᓂᐊᕐᓂᖅ ᓱᓕ ᑲᔪᓯᔪᖅ ᕿᒥᕐᕈᓂᕐᒥ ᐱᓕᕆᔾᔪᑎᒥ–ᑐᙵᕕᓕᕐᒥ ᑐᕌᒐᕐᒥ ᑲᑎᑎᑦᑎᓇᓱᒡᓗᑎᒃ ᑖᒃᑯᓂᙵ ᒪᕐᕉᖕᓂ ᕿᒥᕐᕈᓂᐅᔪᐃᑦ ᐃᑲᕐᕋᖏᓐᓂ ᓴᓂᓕᕇᓕᖅᑎᑦᑎᓗᓂ.

ᐱᓕᕆᔾᔪᑎᓂ ᖃᐅᔨᔭᐅᔪᑦ ᒪᓕᒃᖢᒋ, ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᖏᑦ ᑐᓂᓯᓚᐅᖅᑐᑦ ᐊᑦᑕᕐᓇᖅᑐᓂ–ᑐᙵᕕᓕᕐᓂ ᓂᕿᒃᓴᓂᕐᓂᕐᒧ ᖁᓕᕇᖅᒍᔪᓄᑦ ᐆᒃᑑᑎᐅᔪᓂ ᐃᓂᐅᔪᒧ ᐊᒡᒍᑐᖅᓯᒪᔭᖅ 0+1 (ᐃᒪᐃᓐᓇᕐᒥᑦ) ᐊᑯᑭᑦᑐᑦ ᖃᑕᖅᑲᑕᖏᓐᓂᑦ 2025–ᒥ ᐊᒻᒪᓗ 2026–ᒥ ᖃᐅᔨᓴᖅᓯᓂᕐᒧ ᐅᓂᒃᑳᓕᐊᒥ ᓇᓂᓴᐅᔪᓐᓇᖅᖢᓂ ᐃᓚᒍᑕᖓ 2–ᒥ. ᓂᕿᓂᒃᓴᓂᒃᑐᑦ ᖁᓂᕕᖏᑦ ᐆᒃᑑᑎᐅᔪᑦ ᐅᓂᒃᑳᖅᑐᑦ ᓯᕗᓂᖅᑎᓐᓂ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᓂᒃ ᐃᖅᑲᓗᖁᑕᐅᔪᑦ ᐆᒪᔪᐃᑦ ᐅᖅᑭᖅᒋᓐᓂᖅᒋᑕ ᖃᓄᐃᑦᑑᓂᖏᓐᓂ (B) ᑕᑯᑦᑐᒋᑦ ᐃᖅᑲᓗᒐᓱᖕᓃᑦ ᐊᔾᔨᖏᙱᑦᑕᖏᓐᓂᑦ ᑐᖁᒃᑕᖅᑕᕐᓂᐅᔪᑦ ᖁᓕᕇᖏᓐᓂ (ᐃᖅᑲᓗᒐᓱᖕᓂᖓ ᐊᒃᓯᕈᕐᓂᐅᔪᑦ, F). ᓂᕿᓂᒃᓴᓂᒃᑐᑦ ᖁᓂᕝᕕᖏᑦ ᐆᒃᑑᑎᐅᔪᑦ ᐊᑐᖅᑕᐅᔪᑦ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒥ ᐊᐅᓪᓚᑎᓐᓂᕐᒋᓐᓇ ᐋᖅᑭᒃᓯᐃᖁᑦᑐᒋ ᓂᕈᐊᒐᒃᓴᓂ ᑲᑎᑦᑐᒋ ᐃᖅᑲᓗᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐅᖃᐅᔾᔭᖅᓯᐃᓂᕐᒥ ᐊᒻᒪᓗ ᐊᑐᖅᑯᓯᐅᔪᓐᓂᑦ ᒥᓂᑦᑕᒍᑦ ᑐᕌᖅᑐᒍ.

ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᖏᑦ ᐅᖃᐅᔾᔨᖅᓯᐃᓂᖓ ᐃᖅᑲᓗᒃᑕᐅᔪᑦ ᖁᓕᕇᖅᓴᖅᓯᓂᖏᓐᓂ ᐅᑯᓄᖓ 2025–ᒥ ᐊᒻᒪ 2026–ᒥ ᐃᒪᐃᑦᑑᕗᑦ:

[ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᖏᑦ] ᐊᑐᓕᖁᔭᐅᔪᖅ ᐅᑯᓄᖓ 2025 ᐊᒻᒪ 2026:

> *ᑐᕌᖅᑕᐅᔪᖅ ᐊᑐᖅᑕᐅᔪᖅ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᖅᓃᑦ B_{lim} ᐊᑦᑎᒃᑐᓪᓚᕆᐅᔪᖅ (<1%), ᐊᒻᒪᓗ ᑕᐃᒪᐃᓐᓇᓴᕐᓂᖓ F_{lim} ᑐᕌᒐᖃᖅᑐᑦ ᑐᖔᓂ 30% ᓇᓕᐊᖕᓂᑐᐃᓐᓇᖅ ᐊᖑᓂᒃᑕᐅᔪᑦ ᑐᖔᓂ 90% ᒫᓐᓇ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᓂ.*
>
> *ᑭᒡᓕᓯᐊᖅᑏᑦ ᑲᑎᒪᔩᑦ ᐊᑐᓕᖁᔨᓯᒪᔪᑦ ᑕᒪᒃᑯᐊ ᐊᖑᓂᒃᑕᐅᔪᑦ ᐅᖓᑕᐅᖅᓴᙱᑦᑐᑎᒃ 90%–ᖓᓂᒃ ᒫᓐᓇ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ.*

ᑲᑎᑦᑐᒋ ᐃᖅᑲᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᑐᓕᖁᔭᐅᔪᖅ ᐋᖅᑭᒃᓯᖅᑕᐅᓯᒪᓪᓗᓂ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᓐᓂ ᑐᙵᕕᖃᓚᐅᖅᑐᖅ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐅᔾᔭᓯᖅᑯᔭᓯᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᒥ ᐋᖅᑭᒃᓯᒪᓂᖏᓐᓄᑦ. ᐊᑦᑕᕐᓇᖅᓂᖓ F ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᓂᕐᒋᑦ ᖄᖏᖅᓯᒍᓂ F_{lim} (ᓲᕐᓗ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᓂᕐᒋᑦ ᐃᖅᑲᓗᓕᕆᓂᖅ ᐃᖅᑲᓗᖁᑕᐅᔪᓂᑦ ᐅᓄᓗᐊᖅᑐᓂ ᐃᖅᑲᓗᒐᓱᖕᓂᖓ) 30% ᑐᖔᓃᒃᑯᓂ. ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐅᔾᔭᓯᖅᑯᔭᓯᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᒥ ᐋᖅᑭᒃᓯᒪᓂᖅᒋᑕ ᐃᓗᐊᓂ, ᐊᒃᐸᓱᒃᑐᒪᓇᒃᑐᖅ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᓂᕐᒋᑦ ᑕᐃᓴᐅᔪᖅ 10%–ᒥ, ᐊᒻᒪᓗ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᓂᕐᒋᑦ ᐊᒃᐸᓱᒃᑐᓂ 30% ᐳᓴᓐᑎ–ᒧ ᑎᑭᑦᑐᒍ (ᐃᓚᒍᑕᖓ 3). ᐱᓕᕆᑦᑎᐊᕐᓂᖓᑕ ᓈᓴᐅᑎᖓ (P) ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᖅᑐᖅ 30% ᑐᖔᓃᑦᑐᓂ ᐃᑲᔫᑎᔪᖅ ᐃᖅᑲᓗᖁᑕᐅᔪᓂ ᖃᓄᐃᖏᑎᐊᖅᑐᓂ ᐱᐅᖏᑦᑐᒍ ᑐᕌᖅᑎᑦᑎᓂᕐᒥ ᓄᖅᑲᖅᑎᑦᑎᔪᖅ ᓯᕗᓂᖓᓂ.

ᑐᓴᕋᓱᐊᕐᓃᑦ

ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᓛᑦ (EAGSAC) ᓴᖅᑭᑎᑕᐅᓚᐅᖅᑐᖅ ᐃᓚᒋᔭᐅᓪᓗᓂ ᖃᔅᓯᒪᓚᖕᓂ ᐱᓕᕆᓂᐅᔪᓂ ᖃᐅᔨᒪᖁᓪᓗᒋ ᐃᖃᓗᓕᕆᓂᕐᒧ ᐊᐅᓚᑦᑎᓂᐅᔪᒧ ᐱᓕᕆᔨᖏᑦ ᐃᖅᑲᕐᒥᐅᑕᓂᒃ ᐃᖃᓗᖕᓄᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖏᑦ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0 ᐃᓂᐅᔪᒧ ᐸᓯᔭᐅᔪᓐᓇᖁᓪᓗᒋ, ᓇᓚᐅᑦᑖᒐᒃᓴᐅᖁᓪᓗᒋ ᐊᒻᒪᓗ ᖃᐅᔨᓇᖅᑰᖁᓪᓗᒋ. ᐃᓂᖃᖅᑎᑦᑎᔪᖅ ᐅᖃᖃᑎᒌᓐᓂᖕᓂᕐᒥ ᐊᒻᒪᓗ ᐃᓚᒋᔭᐅᔪᓂ ᐱᕕᒃᓴᖃᖅᑎᑦᑎᓪᓗᓂ ᐅᖃᐅᔾᔪᖅᓯᐃᓂᕐᒥ ᐊᒻᒪᓗ ᐊᑐᓕᖁᔭᐅᔪᓂᑦ ᑐᓂᓯᓂᕐᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒦᒃᑯᓐᓄ ᐊᐅᓚᑦᑎᓂᕐᒥ ᐊᒻᒪᓗ ᐊᑐᐊᒐᓂ ᐃᓚᖃᖅᑐᓂ ᐱᔾᔪᑎᖃᖅᑐᓂ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0 ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑕ ᒥᒃᓵᓄᑦ. ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᓛᑦ ᐃᓚᒋᔭᐅᓂᖅ ᐃᓚᖃᖅᑐᖅ ᐊᐅᓚᑦᑎᖃᑎᒌᓐᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᑦ, ᐱᔪᓐᓇᕐᓂᖃᖅᑐᒥ ᑎᒍᒥᐊᖅᑎᐅᔪᑦ, ᑮᓇᐅᔭᓕᐅᕋᓱᓐᓂᕐᒧᑦ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖏᑦ, ᑮᓇᐅᔭᓕᐅᕋᓱᐊᓂᕐᒧᑦ ᐃᖃᓗᒐᓱᐊᖅᑏᑦ, ᐊᕙᑎᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᑦ ᐊᒻᒪᓗ ᐊᓯᖏᑦ ᒐᕙᒪᒃᑯᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖏᑦ.

ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᓛᑦ ᑲᑎᒪᓚᐅᖅᑐᑦ ᐋᒍᔅᓯ 14, 2024−ᒥ, ᐃᓚᒋᔭᐅᔪᑦ ᐃᓱᒪᒋᔭᖏᑦ ᐅᖃᐅᓯᖃᖁᓪᓗᒋ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᖏᑦ ᐅᖃᐅᔾᔪᖅᓯᐃᓂᖓ ᑭᒡᓕᓯᓂᐊᖅᓯᒪᔪᑦ ᐅᖃᐅᔾᔨᒋᐊᕐᓂᖕᒥ ᐱᔾᔪᑎᖃᖅᑐᓂ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0 ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᑦ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖁᓕᕇᖕᓂᖏᓐᓂ 2025−ᒥ ᐊᒻᒪᓗ 2026 ᐃᖃᓗᒐᓱᖕᓂᐅᔪᒥ. ᑲᑎᒪᓂᕐᒧᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐃᓚᒋᔭᐅᔪᓂᑦ ᑕᑯᓐᓇᖅᑕᖁᔪᑦ ᓇᓂᔭᐅᔪᓐᓇᖅᑐᑦ ᐅᕙᓂ ᐃᓚᒍᑕᖓ 4. ᐃᓚᒋᔭᐅᔪᑦ ᑐᙵᓱᒃᑎᑕᐅᓚᕐᒥᔪᑦ ᑕᑯᓐᓇᖅᑕᖏᓐᓂ ᐅᖃᓪᓚᖁᔭᐅᓪᓗᑎᑦ ᐃᖅᑲᓇᐃᔭᕐᕕᓚᓕᖕᓄᑦ ᑎᑎᖅᑲᑎᒍᑦ.

ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᓛᓂᑦ ᐃᓚᒋᔭᐅᔪᑦ ᐃᑲᔪᖅᓱᐃᓚᐅᖅᑐᑦ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖁᓕᕇᓂ ᑎᑭᐅᑎᓯᒪᓂᖏᓐᓂ 2024−ᒥ ᖁᓕᕇᖑᔪᓂ 16,502.5 t ᐅᕿᖏᓐᓂᕐᒥ, ᐃᓚᖏ ᐱᔪᒪᓂᖃᖅᑐᑎᒃ ᐅᑎᕈᒪᓂᕐᒥ 2022 ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖁᓕᕇᖑᔪᓂ 18,185 t ᐅᕿᖏᓐᓂᐅᔪᒥ. ᑕᒪᐃᓐᓂᐸᑐᒃ ᐃᓚᒋᔭᐅᔪᑦ ᐊᑐᓕᖁᐃᓚᐅᕐᒥᔪᑦ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐋᖅᑭᒃᑕᐅᑐᑎᒃ ᐊᑕᐅᓯᕐᒧ ᐊᕐᕌᒍᒧ (2025) ᑭᒡᓕᓯᓂᐊᖅᓯᒪᔪᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᑏᑦ ᓄᑖᙳᕆᐊᖅᑎᑕᐅᓂᖏᓐᓂ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᖏᑦ ᐱᕕᒃᓴᖅᓯᖁᓪᓗᒋ ᐊᒻᒪᓗ ᐱᐅᓂᖅᓴᒥ ᑐᓴᖅᑎᑦᑎᒋᐊᕐᑐᑎᒃ 2026−ᒥ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᓂ. ᖃᐅᔨᒪᔭᐅᔭᕆᐊᓕᒃ ᓄᑖᙳᕆᐊᖅᓯᒪᔪᖅ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖑᔪᓄᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔩᖏᑦ ᐱᔭᕆᐊᕐᓂᖓ ᐊᔪᕐᓇᑐᐃᓐᓇᓇᐊᖅᑲᖅᑐᖅ ᐃᑐᐃᑦᑐᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ ᔫᓂ 2025 ᐃᖃᓗᖁᑕᐅᔪᓂ ᖃᐅᔨᓴᕐᓂᕐᒧ ᑲᑎᒪᓂᐅᓕᖅᐸ.

ᐃᒪᓐᓈᖅᑰᑏᑦ

ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐃᓱᒪᓕᐅᕆᔭᐅᔪᑦ ᐃᓱᒪᖅᑲᕈᓐᓇᖅᑐᑦ ᖃᔅᓯᒪᓚᖕᓂ ᐱᔾᔪᑎᓂ ᐃᓚᖃᖅᑐᓂ: ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ; ᑭᒡᓕᓯᓂᐊᖅᓯᒪᔪᑦ ᐅᖃᐅᔾᔨᒋᐊᕐᓂᕐᒥ; ᑮᓇᐅᔭᓕᕆᓂᕐᒧᑦ ᐊᑦᑐᖅᑕᐅᔪᑦ; ᐊᐅᓚᑦᑎᖃᑎᒌᓐᓂᕐᒧᑦ ᐊᒻᒪ ᑎᒍᒥᐊᖅᑎᐅᖅᑕᐅᔪᑦ ᑕᑯᓐᓇᖅᑕᖏᑦ; ᓄᓇᑖᕈᑎᒍᑦ ᐱᓕᕆᐊᖑᒋᐊᓃᑦ ᐊᒻᒪ ᓯᓚᕐᔪᐊᒥ ᐃᓱᒪᒋᔭᐅᓂᖏᓐᓄᑦ.

ᓇᓛᖅᑮᕋᔅᓴᐅᔪᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐅᖃᐅᔾᔨᒋᐊᖅᑎᓪᓗᒋ ᐊᑎᑦᓕᒋᐊᖅᑐᓂᑦ ᐃᖃᓗᒃᑕᐅᔪᓂᑦ ᖁᕝᕙᓯᓐᓂᖏᓐᓄᑦ:

- ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᓂ ᐊᑐᓕᖅᑎᑦᑎᓂᕐᒥ ᑭᒡᓕᓯᓂᐊᖅᓯᒪᔪᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᑎᓂ ᒪᓕᒃᑐᓂ, ᐊᑐᓕᖁᐃᓪᓗᑎᒡᓗ ᐃᖃᓗᒃᑕᐅᔪᑦ 90%−ᓂ ᓄᓇᕐᔪᐊᒥ ᓂᕿᒃᓴᓂᒃᑕᐅᔪᑦ ᖁᓕᕇᖏᓐᓂ ᖂᖏᖅᓯᖏᓪᓗᑎᒃ ᐃᓂᐅᔪᒧ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0+1 (ᐃᒪᐃᓐᓇᕐᒥᑦ) 33,305 t ᐅᕿᖏᓐᓂᓕᖕᒥ. ᑖᓐᓇ

ᐃᒪᐃᑦᑎᑦᑎᕗᖅ ᓄᓇᕐᔪᐊᒥ ᓂᕿᒃᓴᓂᒃᑕᐅᔪᑦ ᖁᓕᕇᖏᑦ 29,970 t ᐅᖀᖏᓐᓂᓕᖕᒥ, ᐊᒻᒪᓗ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ 14,985 t ᑲᓇᑕᒧ ᑐᓂᔭᐅᔪᒧ.

- ᐱᓯᒪᓗᒍ ᐊᖅᖀᒎᓚᐅᖅᑐᒧᑦ ᑲᑎᑦᑐᒋᑦ ᐃᖃᓗᑦᑕᐅᔪᓐᓇᖅᑐᑦ (ᓲᕐᓗ, 16,502.5 t).
- ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᓂ ᐊᑦᑎᒃᓯᒋᐊᖅᑎᑦᑎᓗᓂ ᐊᑐᓕᖅᑎᑦᑎᓂᕐᒥ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᐅᖃᐅᔾᔪᖅᓯᐃᓂᖓ ᐅᖃᐅᔾᔨᕆᐊᕐᓂᖏᑕ ᐊᓯᖏᓐᓂ.

ᒪᓕᒃᓗᒍ ᑎᑎᕋᖅᓯᒪᔪᖅ 15, ᐃᓚᖓ 3.4 ᑕᐃᑲᓂ *ᓄᓇᕗᒻᒥ ᐊᖏᖃᑎᒌᒍᑦ*, ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒥ ᕿᓂᖅᑐᒃ ᐅᖃᐅᔾᔨᐊᕈᑎᒥᒃ ᑖᒃᑯᓇᙵᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓂᑦ (NWMB) ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᔪᒃᓴᒥᒃ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᐅᑯᐊ:

1. ᑖᓐᓇ ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᒪᒃᑯᓄᖓ ᐊᑯᑭᑦᑐᒥ ᖃᓕᕋᓛᑦ ᑕᐃᕙᓂ ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0 (ᐃᒪᕙᖕᒥ);
2. ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᓇᓃᓐᓂᖏᑦ ᑖᒃᑯᓇᓂ 0A ᐊᒻᒪᓗ 0B; ᐊᒻᒪᓗ
3. ᐃᓕᐅᖃᖅᑕᐅᓂᖏᑦ ᑑᑕᖏᑦ ᐅᒥᐊᕐᔪᐊᓄᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᓄᓇᖅᐸᓯᒻᒥ ᑑᑕᖏᑦ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ.

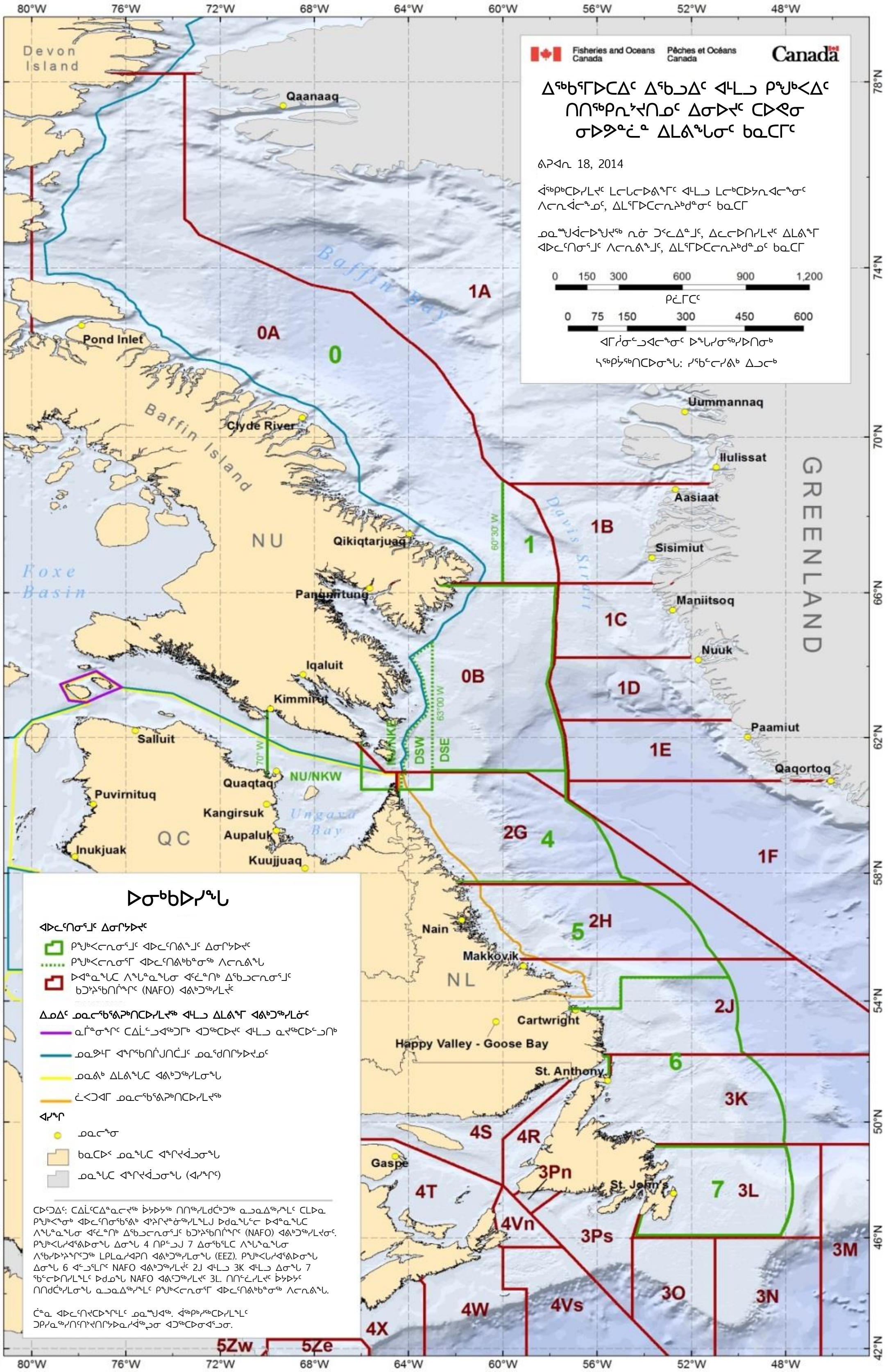
ᐱᕙᕇᑕᖅᑕᖓ ᐆᒪ: ᐊᓕᒃᓯᔅ ᐳᑦ, ᐃᖃᓗᓕᕆᓂᕐᒧ ᐊᐅᓚᑦᑎᔨᓄᑦ (FM) ᐆᒪᔪᓕᕆᔨ, ᔨᕝ ᐋᑕᒻ, ᐊᕙᒃᑐᖅᓯᒪᔪᓂ ᐃᖃᓗᓕᕆᓂᕐᒧ ᐊᐅᓚᑦᑎᔨᓄᑦ ᐱᓕᕆᔨᒻᒪᕆᒃ, ᐊᒻᒪᓗ ᑭᕙᓐ Hᐃᔾᔨ, ᖃᐅᔨᓴᖅᑎᓄ ᑭᓪᓕᓯᓂᐊᖅᑎ, ᐅᑭᐅᖅᑕᖅᑐᒥ ᐊᕙᒃᑐᖅᓯᒪᓂᔪᓂ, ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᓂᕐᒧᓪᓗ ᑲᓇᑕᒥ.

ᐅᓪᓗᖅ: ᐋᒍᔅᓯ 28, 2024

ᐃᓚᒍᑕᖏᑦ

ᐃᓚᒍᑕᖓ 1 – ᓄᓇᙳᐊᖅ ᐃᖃᕐᒥᐅᑕᐃᑦ ᐃᖃᓗᐃᑦ ᐊᒻᒪᓗ ᑭᖑᒃᐸᐃᑦ ᑎᑎᖅᑭᓇᔾᔪᑎᓄᑦ ᐃᓂᐅᔪᑦ ᑕᐅᕙᓂ ᓂᐅᕗᓐᓛᓐ ᐃᒪᕙᖕᒐᓂᑦ ᑲᓇᑕᒥᑦ
ᐃᓚᒍᑕᖓ 2 – ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ
ᐃᓚᒍᑕᖓ 3 – ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᐅᔾᔨᕐᑯᔨᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐋᖅᑭᒃᓯᒪᓂᖏᑦ ᖃᐅᔨᓴᕐᓂᖅ ᑭᓪᓕᓯᓂᐊᕐᓂᕐᒧ ᑲᑎᒪᔨᖏᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ (SCS) 24-13
ᐃᓚᒍᑕᖓ 4 – ᑲᑎᒪᓂᕐᒧᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ ᑲᓇᕐᓇᖓᓂ ᐃᖃᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᓛᑦ ᐃᓕᕆᔭᐅᔪᑦ ᑕᑯᓐᓇᖅᑕᖏᑦ 2025 ᐊᒻᒪᓗ 2026 ᐊᑯᑭᑦᑐᑦ ᖃᓕᕋᓕᖏᓐᓂ ᑲᑎᓪᓗᒋ ᐃᖃᓗᒃᑕᐅᔪᓐᓇᖅᑐᑦ

ᐃᓚᒍᑕᖕᒐ 1



ᐃᖅᑲᖁᒥᐅᑕᐃᑦ ᐃᖃᓗᐃᑦ ᐊᒻᒪᓗ ᑭᖑᒃᐸᐃᑦ ᑎᑎᖅᑭᓇᔾᔪᑎᓄᑦ ᐃᓂᐅᔪᑦ ᑕᐅᕙᓂ ᓂᐅᕗᓐᓃᓐ ᐃᒪᕙᖓᓂᑦ ᑲᓇᑕᒥᑦ

ᕕᐳᐊᕆ 18, 2014

ᐋᖅᑭᒃᑕᐅᓯᒪᔪᑦ ᒪᓕᒐᓕᐅᕙᖕᒥᑦ ᐊᒻᒪᓗ ᒪᓕᒃᑕᐅᔭᕆᐊᓕᖓᓂᑦ ᐱᓕᕆᐋᓕᖓᓄᑦ, ᐃᒪᕐᒥᐅᑕᓕᕆᔩᒃᑯᓐᓂᑦ ᑲᓇᑕᒥ

ᓄᓇᙳᐋᓕᐅᖑᔪᖅ ᕆᓃ ᑐᑉᓚᐃᓐᒧᑦ, ᐃᓚᓕᐅᑎᓯᒪᔪᑦ ᐃᒪᕙᖕᒥ ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᓕᕆᕙᖕᒧᑦ, ᐃᒪᕐᒥᐅᑕᓕᕆᔩᒃᑯᓐᓄᑦ ᑲᓇᑕᒥ

ᓴᖅᑮᔭᖅᑎᑕᐅᓂᖓ: ᓯᖃᓪᓕᓯᕙᒃ ᐃᓗᓕᒃ

ᑕᐅᑦᑐᐃᑦ: ᑕᐃᒻᑕᐃᓐᓇᓕᔪᖅ ᐆᔭᐅᔭᖅ ᑎᑎᖅᓯᒪᑯᑖᒃᑐᖅ ᓇᓗᓇᐃᖅᓯᖓᓪ ᑕᒪᐅᓇ ᑭᖑᒃᐸᖓᓂᒃ ᐊᐅᓚᕐᑎᓂᖅᑲᖅᕙᒃ ᐊᔾᔨᙵᓐᓂᖅᓯᒪᖓᒍ ᐅᑯᓇᖓᓪᓚ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑦᔫᓐᑎᒃ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ (NAFO) ᐊᕙᒃᑐᖅᓯᒪᔪᓂᑦ. ᑭᖑᒃᐸᓚᓯᐊᖅᕕᐅᓂᖓ ᐃᓂᖓ 4 ᑎᑭᓪᓗᒍ 7 ᐃᓂᖅᑲᖅᒪᑕ ᐱᖓᖕᓇᖓᓂ ᐱᖃᓯᐅᔾᔨᖕᖏᑦᑐᖅ ᒪᑭᒪᓇᓯᐊᕈᑎ ᐊᕙᒃᑐᖅᓯᒪᓂᖓ (EEZ). ᑭᖑᒃᐸᓚᓯᐊᖅᕕᐅᓂᖓ ᐃᓂᖓ 6 ᐊᓪᓗᖅᒪᒋᑦ NAFO ᐊᕙᒃᑐᖅᓯᒪᔩᑦ 2J ᐊᒻᒪᓗ 3K ᐊᒻᒪᓗ ᐃᓂᖓ 7 ᖃᓪᓕᐅᑎᓯᒪᖓᒪᑦ ᐅᑯᓄᖓ NAFO ᐊᕙᑦᑐᖅᓯᒪᔪᑦ 3L. ᑎᑎᓪᓛᓯᒪᔪᑦ ᐆᔭᐅᔭᑦ ᑎᑎᖁᑖᒃᓯᒪᓂᖓ ᓇᓗᓇᐃᖅᓯᖓᓪ ᑭᖑᒃᐸᓕᕆᓂᕐᒥ ᐊᐅᓚᕐᑎᕙᒃᑲᓂᖅ ᐱᓕᕆᕙᖓ.

ᑖᓐᓇ ᐊᐅᓚᕐᑎᔪᑕᐅᖏᒻᒪᑦ ᓄᓇᙳᐊᖅ. ᐋᖅᑭᒃᓯᖅᑕᐅᓯᒪᖓᒪᑦ ᑐᑭᓯᓇᖅᓯᑎᑦᑎᔾᔪᑎᒋᔭᐅᓇᓱᐋᖅᖢᓂ ᐊᑐᖅᑕᐅᓂᐊᕐᓗᓂ.

ᖃᓕᕋᓛᖕᓂᒃ ᓇᒦᓐᓂᖓ 0+1 (ᐃᒪᐃᓐᓇᕐᒥᑦ) ᐅᖃᐅᔾᓯᕆᐊᕈᑎᑦ ᔫᓂ 2024 ᑕᒪᑯᖓ 2025-2026

ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᔪᑦ

ᐊᑐᓕᕐᑯᔭᐅᔪᖅ ᐅᑯᓄᖓ 2025 ᐊᒻᒪ 2026

ᑐᖀᖅᑕᐅᔪᖅ ᐊᑐᖅᑕᐅᔪᖅ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖃᕐᓃᑦ B_{lim} ᐊᑦᑎᒃᑐᓪᓚᓇᐅᔪᖅ (<1%), ᐊᒻᒪᓗ ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ F_{lim} ᑐᖀᒐᖃᖅᑐᑦ ᑐᙱᓂ 30% ᓇᓕᐊᖕᓂᑐᐃᓐᓇᖅ ᐊᖑᓂᒃᑕᐅᔪᑦ ᑐᙱᓂ 90% ᒦᓐᓇ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔫᓐᓇᖅᑐᓂ.

ᑭᒡᓕᓯᐊᕐᕖᑦ ᑲᑎᒪᔩᑦ ᐊᑐᓕᕐᑯᔨᓯᒪᔪᑦ ᑕᒪᒃᑯᐊ ᐊᖑᓂᒃᑕᐅᔪᑦ ᐅᖓᑕᐅᔾᙱᓪᓗᑎᒃ 90%–ᖓᓂᒃ ᒦᓐᓇ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔫᓐᓇᖅᑐᑦ.

ᑲᓇᑕᒥ ᐊᒻᒪᓗ ᑕᓐᒪᐅᒃᒥ (ᐱᓕᕆᔾᔪᔾᔨᓪᓗᑎᒃ ᑲᓛᓖᑦ ᓄᓈᓂ) ᑐᒃᓯᕋᖅᓯᒪᔪᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐊᑐᐃᓐᓇᕈᐃᓗᑎᒃ ᐊᑕᖕᒌᓯᒪᔪᒥᒃ ᕿᒥᕐᖁᓂᕐᒥᒃ ᖃᓄᐃᓕᖓᓂᖏᓐᓄᑦ ᐊᒻᒪᓗ ᐃᓕᖅᑯᓯᐅᔪᑦ ᑲᑎᓪᓗᒋᑦ ᐱᑕᖃᕕᐅᔪᑦ ᑕᐅᕙᓂᓗᒃᑖᖅ ᐱᑕᖃᕕᐊᓂ. ᐱᑕᖃᕕᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ ᕿᒥᕐᕈᔭᐅᓪᓗᐊᖅᑐᑦ ᐃᓱᒪᒋᓪᓗᒋᑦ ᐊᐅᓚᕐᑎᔾᔪᑕᐅᔭᕆᐊᓖᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ ᑲᔪᓲᓐᓇᕈᓐᓇᖅᑯᓪᓗᒋᑦ ᐊᒻᒪᓗ ᐅᖃᐅᔾᓯᕆᐊᕈᑎᑦ ᑐᓴᖅᑎᑦᑎᔾᔪᑕᐅᔪᑦ ᐊᔾᔨᒌᓪᓗᐊᖅᑐᑦ ᑳᒃᑯᓄᖓ ᐅᐊᓐᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐅᔾᔨᕈᓱᒃᑯᔨᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐋᖅᑭᒃᓯᒪᓂᕐᒥᑦ.

ᑲᑎᒪᕐᔪᐊᕐᓂᕐᒧᑦ ᑐᙵᕕᖕᒋᑦ	ᖃᓄᐃᓕᖓᓂᖏᑦ	ᐅᖃᐅᓯᒃᓴᖅ
ᐅᖅᑎᓪᓗᒋᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑕᐃᒪᐃᓕᖓᑏᓐᓇᕐᓗᒋᑦ ᑕᐃᑲᓂ Bmsy	🟢	B > Bmsy
ᐱᑕᖃᕈᓐᓃᖅᑎᓪᓗᒋᑦ ᐃᖃᓗᒐᓱᓗᐊᕐᓃᑦ (ᐱᑕᖃᕕᐅᔪᒥ)	🟢	F < Flim
ᐱᑕᖃᕈᓐᓃᖅᑎᓪᓗᒋᑦ ᐃᖃᓗᒐᓱᓗᐊᕐᓃᑦ (ᐆᒪᓂᖅᑲᖅᑐᑦ ᓇᔪᒐᕐᒥᓂᑦ)	⚪	TCI ᓇᓗᓇᐃᔭᖅᑕᐅᓯᒪᙱᑦᑐᑦ
ᐊᑐᓗᓂ ᐊᑦᑕᓇᐃᖅᓯᒪᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓂᖓ	🟢	Blim ᐊᒻᒪᓗ Flim ᓇᓗᓇᐃᔭᖅᑕᐅᓯᒪᙱᑦᑐᑦ
ᐊᑦᑎᓪᓕᕆᐊᓗᒍ ᐊᑦᑕᓇᕈᓐᓇᖅᑐᓄᑦ ᐊᑦᑕᖅᑕᐅᔪᑦ ᐆᒪᔪᓄᑦ ᐃᒪᕐᒥᐅᑕᓂᑦ ᐊᒻᒪ ᐊᕙᑎᖓᓄᑦ	🟡	ᑐᑭᒧᐊᒃᑎᑕᐅᔪᑦ ᐃᖃᓗᓕᕆᓃᑦ VME ᒪᑐᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᑦ, ᐊᑑᑎᖃᑦᑎᐊᖅᑐᑦ ᐊᖑᓂᒃᑕᐅᔪᑦ ᒪᓕᒐᕐᒥᑦ ᓇᓗᓇᖅᑐᑦ
ᔭᒐᐃᑦᑕᐃᓕᒪᓃᑦ ᑕᕆᐅᒥ ᐆᒪᓂᖃᖃᖅᑮᑭᓘᔭᖅᑐᑦ	⚪	ᖃᐅᔨᓴᖅᑕᐅᔪᓐᓇᙱᑦᑐᑦ

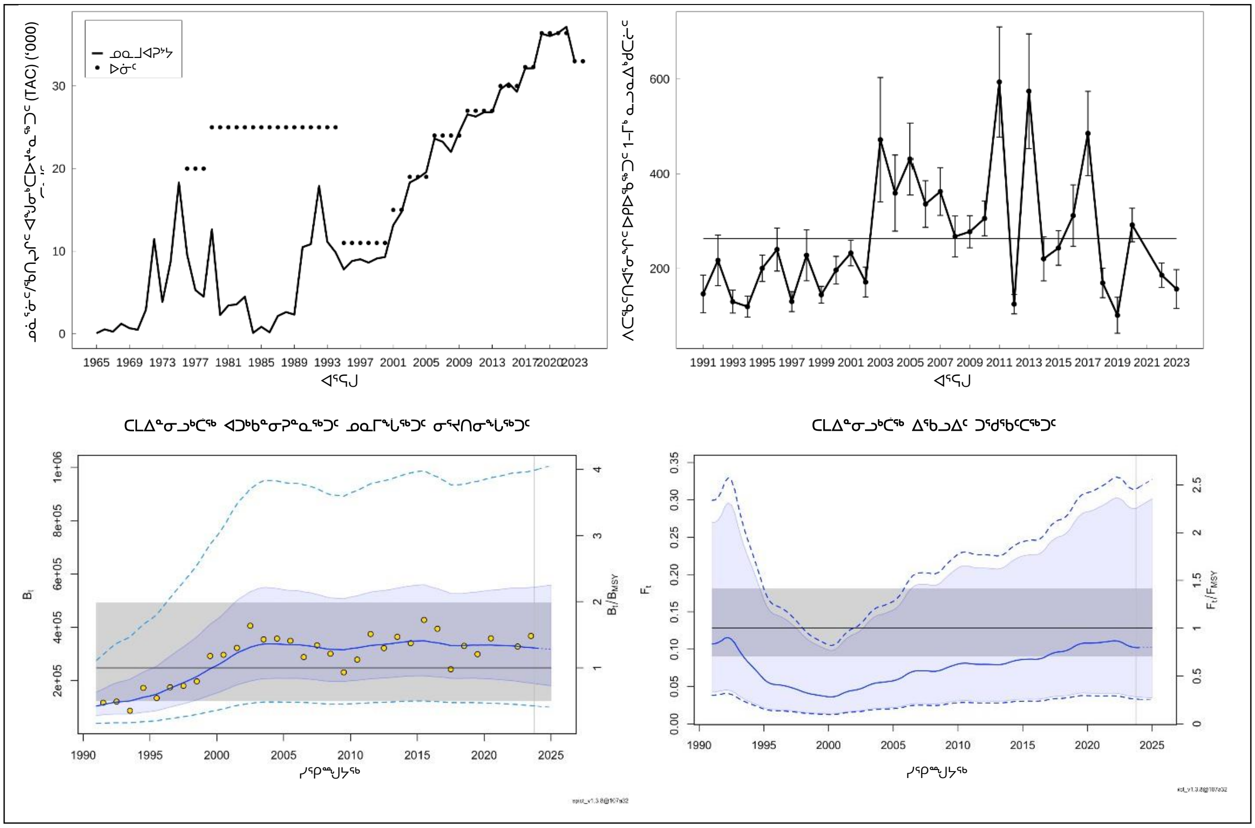
🟢 ᖃᓄᐃᙱᑦᑐᖅ
🟡 ᓇᓗᓇᐃᖅᓯᒪᓪᓚᕆᙱᑦᑐᖅ
🔴
⚪ ᐱᔭᕇᖅᓯᒪᙱᑦᑐᖅ

ᐊᐅᓚᕐᑎᕕᒋᔭᐅᔪᖅ

ᑲᓛᓖᑦ ᓇᑖᕐᓇᖏᓐᓂᒃ ᐱᑕᖃᕕᐅᔪᑦ ᐊᓪᓚᐅᓂᖅᓴᑦ ᐃᓂᑦ 0+1 (ᐃᒪᐃᓐᓇᕐᒥ) ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᐅᓄᕐᓂᖅᓴᑦ ᓇᓗᓇᖅᑐᑦ ᓇᓃᓐᓂᖏᑦ ᑕᐅᕝᕙᓇᓗᒃᑖᖅ ᐅᐊᓐᓇᔫᖓᓗᒍᑦ ᐱᓇᓐᓇᖅᐸᓯᒃᑯᑦ ᐊᑦᓛᓐᑎᒃᒥ.

ᐃᖃᓗᖕᒥᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ

ᕿᑎᖅᑲᖅᓯᒪᔪᑦ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙱᖅᑐᑦ ᓂᕙᑎᓂᙱᖅᑐᑦ ᐅᖓᑖᓂᑦᑐᑦ B_{msy} (B/B_{msy} = 1.3) ᐊᒻᒪ ᑐᙱᓗᐊᕋᔭᕐᓂᖓ B_{lim} ᑐᙱᓃᑦᑐᖅ 1%. ᐃᖃᓗᓕᕆᓂᖅ ᑐᕐᑯᑕᐅᒃᑕᖅᑐᑦ ᑐᙱᓃᑦᑐᑦ F_{msy} (F/F_{msy} = 0.78) ᐊᒻᒪᓗ ᖁᖓᑖᓃᑦᑐᐃᓐᓇᓇᐊᖃᕐᓂᖓ F_{lim} 34%–ᖑᔪᖅ.



ᓇᓗᓇᐃᕐᕕᖓᓂ ᓇᑭᙶᕐᓂᖓᓂ

B_{lim} 30%–ᖑᔪᖅ B_{msy} ᐊᒻᒪᓗ F_{lim} ᐃᒪᐃᑦᑑᓪᓗᓂ F_{msy} (SCS 04/12).

ᓇᓚᐅᑦᑖᕈᑎᐅᔪᑦ

ᐊᑯᓐᓂᖅᓱᖅᑐᖅ–ᐊᑯᓂᐅᓂᖓᓄᑦ ᑐᕌᒐᐅᔪᑦ ᑲᔪᓯᐊᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᐊᕐᕌᒍ 2026–ᒧᑦ ᐊᖑᓂᒃᑕᐅᔪᓄᑦ ᐋᖅᑭᙳᐊᖅᑕᐅᓯᒪᔪᑦ = ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAC) 33 305 ᑕᓐᓯᑦ 2024–ᒧᑦ. ᐲᔭᖅᑕᐅᖏᓐᓇᕐᓂᖏᑦ ᐊᑐᓕᖅᑎᑕᐅᓯᒪᔪᑦ ᑕᕝᕙᙵᑦ 2025-2026 ᐊᒥᓱᒐᓚᖕᓂ ᖁᕝᕙᓯᓕᓅᓂ ᑕᐃᑲᙵᑦ F (F=0, $F_{\text{ᖃᓄᐃᓕᖓᓕᕐᓂᖏᑦ}}$, 75% F_{msy}, 85% F_{msy} ᐊᒻᒪᓗ F_{msy}) ᐅᕝᕙᓘᓐᓃᑦ (ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ 90% ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ). ᐃᓱᐊᓂ ᑐᕌᒐᖃᕐᕕᐅᑉ, ᐅᓗᕆᐊᓇᕈᓐᓇᕐᓂᖓ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖔᖅᑐᑦ ᓂᕐᔪᑎᓂᖔᖅᑐᑦ ᑐᖔᓃᓕᕐᑐᑎᒃ B_{lim} ᐃᒪᐃᑦᑑᓚᐅᖅᑐᖅ 1% ᐱᓕᕆᐊᖑᔪᒃᑖᓂ.

ᑕᒪᑐᒧᖓ $F_{\text{ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓄᑦ}}$ ᑐᕌᖅᑕᐅᔪᑦ, ᐃᒪᐃᓐᓇᔭᕐᓂᖓ $F > F_{lim} = F_{msy}$ ᐊᑐᖅᑎᓪᓗᒍ 2025-2026 ᐃᒪᐃᑦᑑᓚᐅᖅᑐᖅ 34%, ᐊᒻᒪᓗ ᐱᑕᖃᖅᐸᑦ 2/3 F_{msy} ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ 23%–ᖑᓚᐅᖅᑐᖅ. 75%–ᖓᓂ F_{msy}, ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ ᑕᒪᓐᓇ $F > F_{lim}$ 30%–ᖑᓯᒪᔪᖅ. ᑐᕌᖅᑕᐅᔪᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ 85%–ᖑᔪᖅ F_{lim}, ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓᓗ ᑖᓐᓇ $F > F_{lim}$ ᐃᒪᐃᓯᒪᓪᓗᓂ 39% ᐊᒻᒪᓗ ᑕᒪᑐᒧᖓ F_{msy} ᑐᕌᖅᑕᐅᔪᑦ, ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ ᐱᒋᐊᓪᓚᒃᓯᒪᔪᖅ 50%-ᒧᑦ. ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖔᖅᑐᑦ ᓂᕐᔪᑎᓂᖔᖅᑐᓄᑦ ᑐᕌᒐᐅᔪᑦ, ᐊᑕᖏᖅᖢᒋᑦ ᖃᓄᐃᓕᖓᓂᙳᐊᓂ 2025-2026–ᒧᑦ ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖔᖅᑐᑦ ᓂᕐᔪᑎᓂᖔᖅᑐᑦ ᑐᖔᓄᐊᕋᔭᕐᓂᖏᑦ B_{lim} ᑐᖔᓃᓚᐅᖅᑐᖅ 1%. ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ ᑕᒪᒃᑯᐊ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖔᖅᑐᑦ ᓂᕐᔪᑎᓂᖔᖅᑐᑦ 2026–ᒥ ᑐᖔᓃᑦᑐᑦ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖔᖅᑐᑦ ᓂᕐᔪᑎᓂᖔᖅᑐᑦ 2024–ᒥ ᐊᑯᓐᓂᖓᓃᖢᑎᒃ 19 ᐊᒻᒪᓗ 70%–ᖑᓪᓗᓂ ᐊᑕᖏᖅᖢᒋᑦ ᑐᕌᒐᐅᔪᓄᑦ.

ᑐᕌᒐᐅᔪᑦ ᐊᖑᓂᒃᓂᕐᒧᑦ 2024 = 33305 ᑕᓐᓂᑦ		
ᐊᕐᕌᒍ	ᓴᖅᑭᔭᖅ ('000 ᑕᓐᓂᑦ)	ᑐᕌᖅᑕᐅᔪᑦ ᐊᒃᑐᐊᓂᓖᑦ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖕᓛᖅᑐᑦ ᓂᕐᔪᑎᓂᖕᓛᖅᑐᓂ (B/Bmsy) ᐊᑯᓐᓂᖅᓱᖅᑐᖅ (80%CL)
F =0		
2024	33.3	1.3 (0.91-1.84)
2025	0	1.28 (0.89-1.85)
2026	0	1.4 (1.02-1.92)
Fᖃᓄᐃᓕᖓᓕᕐᓂᖏᑦ = 0.102		
2024	33.3	1.3 (0.91-1.84)
2025	32.33	1.28 (0.89-1.85)
2026	32.04	1.27 (0.87-1.86)
2/3Fmsy= 0.085		
2024	33.3	1.3 (0.91-1.84)
2025	27.23	1.28 (0.89-1.85)
2026	27.39	1.28 (0.91-1.88)
75%Fmsy = 0.096		
2024	33.3	1.3 (0.9-1.85)
2025	30.51	1.28 (0.89-1.86)
2026	30.4	1.26 (0.89-1.87)
85% Fmsy = 0.109		
2024	33.3	1.3 (0.91-1.84)
2025	34.42	1.27 (0.89-1.85)
2026	33.91	1.26 (0.86-1.85)
Fmsy = 0.128		
2024	33.3	1.3 (0.91-1.84)
2025	40.21	1.28 (0.89-1.85)
2026	38.92	1.24 (0.83-1.84)
ᑲᑎᒍᒃᑕᖅᒍᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ = 33 305		
2024	33.3	1.3 (0.91-1.84)
2025	33.3	1.28 (0.89-1.85)
2026	33.3	1.27 (0.86-1.85)
90% ᑲᑎᒍᒃᑕᖅᒍᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ = 29 975		
2024	33.3	1.3 (0.91-1.84)
2025	29.97	1.28 (0.89-1.85)
2026	29.97	1.28 (0.88-1.86)

	ᓴᖅᑮᔭᖅ ('000 ᑕᓐᓂᑦ)		P (F> Flim)			P (B<Blim)			P (B>Bmsy)			P (B2026 < B2024)
ᐊᖑᓂᒃᑕᐅᔪᑦ 2024 = 333	2025	2026	2024	2025	2026	2024	2025	2026	2024	2025	2026	
F=0	0	0	34%	<1%	<1%	<1%	<1%	<1%	83%	81%	91%	19%
Fᖃᓄᐃᓕᖓᓕᕐᓂᖏᑦ	32.33	32.04	34%	34%	34%	<1%	<1%	<1%	83%	81%	79%	60%
2/3 Fmsy	27.23	27.39	34%	23%	23%	<1%	<1%	<1%	83%	81%	81%	53%
75% Fmsy	30.51	30.4	34%	30%	30%	<1%	<1%	<1%	83%	81%	80%	58%
85% Fmsy	34.42	33.91	34%	38%	39%	<1%	<1%	<1%	83%	81%	78%	63%
Fmsy	40.21	38.92	34%	50%	50%	<1%	<1%	<1%	83%	81%	76%	70%
ᑲᑎᒍᒃᑖᖅᒍᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ	33.3	33.3	34%	36%	37%	<1%	<1%	<1%	83%	81%	79%	62%
90% ᑲᑎᒍᒃᑖᖅᒍᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ	29.97	29.97	34%	29%	29%	<1%	<1%	<1%	83%	81%	80%	57%

ᖃᐅᒡᓴᖅᑕᐅᓂᖓ

ᐋᖅᑭᒃᓱᐃᔾᔪᑏᑦ ᑲᔪᓯᕗᓪᓕᐊᖏᓐᓇᖅᑐᓄᑦ (SPiCT) ᐊᑐᖅᑕᐅᓯᒪᔪᖅ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᓗᒋᑦ ᑖᒃᑯᐊ ᐱᑕᖃᕐᕕᐅᔪᑦ. ᐱᕝᕚᐅᔪᑦ ᑕᒪᑐᒧᖓ ᐱᓕᕆᔾᔪᑎᒧᑦ ᐱᖃᓯᐅᔾᔨᔪᑦ ᓄᓈᕐᓂᕐᒥ ᑎᑎᖅᑲᖁᑏᑦ ᐊᒻᒪᓗ ᐊᑐᖅᑕᐅᖓᔪᒃᑐᑦ ᓇᓗᓇᐃᔭᐅᑏᑦ ᑖᒃᑯᓇᙵᓂᑦ ᐱᕝᕚᐅᔪᓐᓇᖅᑐᓂᒃ ᐱᑕᖃᕐᕕᐅᔪᓂ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖕᓛᖅᑐᑦ ᓂᕐᔪᑎᓂᖕᓛᖅᑐᓂ ᑲᑎᑕᐅᓯᒪᔪᑦ ᖃᐅᔨᓴᕐᓂᕐᒥ ᑎᑎᖅᑲᖁᓂ.

ᑭᖑᓪᓕᕐᒥᑦ ᖃᐅᔨᓴᕐᓂᐅᓛᖅᑐᖅ ᓂᕆᐅᒋᔭᐅᔪᖅ 2026.

ᐃᓄᓐᒧᑦ ᐊᑦᑐᖅᑕᐅᔪᑦ

ᐃᖃᓗᓕᕆᓂᐅᒐᔪᑦᑐᑦ ᐊᑦᑐᐊᓂᓖᑦ ᑐᖁᒃᑲᑦᑕᖅᓯᒪᔪᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓕᑎᓪᓗᒋᑦ. ᐊᓯᖏᑦ ᓇᑭᖕᒐᕝᕕᖏᑦ (ᒎᖅᑐ. ᓱᑲᓐᓇᖅᑐᑦ, ᐅᒥᐊᖅᔪᐊᓕᕆᓂᖅ, ᐅᖅᓱᐊᓗᒃᓴᓯᐅᖅᓂᖅ) ᑎᑎᕋᖅᑕᐅᓯᒪᖏᑦᑐᑦ.

ᐆᒪᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐊᒻᒪ ᐊᕙᖓᓄᑦ ᖃᓄᐃᓘᖃᑦᑕᐅᑎᓂᖏᑦ

ᓇᓗᓇᐃᖅᓯᒪᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᕿᒥᕐᕈᔭᐅᔪᖃᓚᐅᖏᑦᑐᖅ ᑕᕝᕙᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ.

ᐆᒪᓂᖃᖅᑐᑦ ᓇᔪᒐᕐᒥᓂᑦ ᑲᔪᓰᓐᓇᕈᓐᓇᖅᑐᒃᑯᑦ ᐊᖑᓂᒃᑕᐅᔪᓂ

ᐊᒃᑐᐃᓃᑦ ᐊᑦᑎᓛᑦ ᐃᖃᓗᒐᓱᖕᓂᕐᒥᑦ ᑕᒪᑐᒧᖓ VME–ᓂᑦ ᐊᓪᓚᐅᓂᖅᓴᓂ ᐃᓂᓂ 0 ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᖅ 2016–ᒥ. ᐱᖓᓱᑦ ᐃᓃᑦ ᑎᒃᑯᐊᖅᑕᐅᓯᒪᔪᑦ ᑕᕆᐅᒥ ᕿᓛᕝᕕᐅᔪᑦ, ᐱᖃᓯᐅᔾᔨᙱᑦᑐᑦ ᐃᖅᑲᖓᓂ ᑕᑯᖃᑦᑕᐅᑎᓂᐅᕙᒃᑐᑦ ᐃᖃᓗᓕᕆᓂᕐᒥ: ᑎᔅᑰ ᕚᓐ, ᑕᐃᕕᔅ ᐃᑭᖓ ᐊᒻᒪᓗ ᕼᐊᑕᓐ ᑲᖏᖅᖢᐊ. ᐃᓃᑦ ᐊᓪᓚᐅᓂᖅᓴᓂ ᐃᓂᒥ 1–ᒥ ᑐᒪᔭᐅᓯᒪᔪᑦ ᐃᖅᑲᖓᓂ ᐃᖃᓗᖓᓯᖖᓂᕐᒧᑦ ᓴᐳᔾᔭᐅᓯᒪᖁᓪᓗᒋᑦ ᐃᖅᑲᖓᓂ ᓇᔪᒐᐅᔪᑦ.

ᑲᓛᓖᑦ ᓇᑖᕐᓇᖏᑦ ᐃᓚᓕᐅᔾᔭᐅᓯᒪᔪᑦ ᐊᖑᓇᓱᒃᐸᒃᑐᓂ ᐊᕕᒃᑐᖅᓯᒪᔪᓂ. ᐱᑕᖃᙱᑦᑐᖅ EPUs ᐅᕝᕙᓘᓐᓃᑦ TCIs ᓇᓗᓇᐃᖅᑕᐅᓯᒪᒧᑦ ᑕᐅᕗᖓ. ᐆᒪᓂᖃᖅᑐᑦ ᓇᔪᒐᕐᒥᓂᑦ ᑲᔪᓰᓐᓇᖁᓪᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓐᓇᖅᓯᒪᙱᑦᑐᑦ. ᐃᖃᓗᒡᔪᐊᑦ ᐱᔭᐅᔪᒪᙱᑦᑑᓪᓗᓂ ᐆᒪᔪᓂ ᐊᔾᔨᒌᙱᐊᕐᔪᒃᑐᓂ ᐃᓱᒫᓘᑕᐅᔪ ᑕᐅᕙᓂ ᐊᓪᓚᐅᓂᖅᓴᒥ ᐃᓂᓂ 0+1 (ᐃᒪᕕᖕᒥ) ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᔾᔪᑕᐅᓪᓗᑎᒃ ᐃᕐᓂᐅᙱᓰᔭᒃᑲᐅᓗᐊᕐᓂᖏᓐᓄᑦ, ᐱᕈᖅᐸᓪᓕᐊᕐᓰᔭᖃᑦᑕᕐᓂᖏᓐᓂᓄᑦ ᐊᒻᒪᓗ ᑭᒡᓕᓕᖕᒥ ᐆᒪᔪᑦ ᓇᔪᒐᖏᑕ ᓯᓚᑎᖏᑕᓗ ᑐᓴᐅᒪᔾᔪᑎᖏᓐᓄᑦ.

ᐃᖃᓗᒐᓱᐊᕐᓂᖅ

ᐃᖃᓗᑦᑕᐅᖃᑦᖅᑐᑦ ᑎᑎᕋᖅᑕᐅᒥᐊᓐᖓᓚᐅᖅᑐᑦ 1965. ᐃᖃᓗᑦᑕᐅᔪᑦ ᖁᕝᕙᓇᐊᓚᐅᖅᑐᑦ ᐱᒋᐊᖅᑐᓂ 1989 ᑎᑭᑦᑐᒍ 1992 ᐱᔾᔪᑎᒋᓪᓗᒍ ᓄᑖᒧᑦ ᑲᓕᒻᒧᑦ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐊᕕᑦᑐᖅᓯᒪᔪᒥᑦ 0B ᐃᓚᐅᖃᑕᐅᓪᓗᑎ ᑲᓇᑕ, ᓄᐊᕙᐃ, ᐅᓛᓵᒃᑯᑦ ᐊᒻᒪ ᕙᐃᑭ ᕿᑭᖅᑕᐃᑦ ᐊᒻᒪ ᐊᖏᓪᓕᒋᐊᖅᑕᐅᓪᓗᓂ ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 1CD ᐃᖃᓗᓕᕆᓂᐅᔪᖅ ᐃᓚᐅᖃᑕᐅᓪᓗᑎ ᔫᕐᕋᓃᓯᒃᑯᑦ, ᓄᐊᕙᐃ ᐊᒻᒪ ᕙᐃᑭ ᕿᑭᖅᑕᐃᑦ. ᐃᖃᓗᑦᑕᐅᔪᑦ ᐊᑦᑎᓪᓕᒋᐊᓚᐅᖅᑐᑦ ᐱᒋᐊᖅᑐᓂ 1992 ᑎᑭᑦᑐᒍ 1995 ᐱᔾᔪᑎᒋᓗᐊᖅᑐᒍ ᐊᑦᑎᓪᓕᒋᐊᕐᓂᖏᓐᓄᑦ ᐱᓇᓱᐊᖅᑕᐅᔪᓄᑦ ᑲᓇᑕᒥᐅᑕᐅᖏᑦᑐᓄᑦ ᐅᒥᐊᕐᔪᐊᓄᑦ ᐅᕙᓂ ᐊᕕᑦᑐᖅᓯᒪᔪᖅ 0B. 1995–ᒥᓂᑦ ᐊᖑᓂᒃᑕᐅᔪᑦ ᖃᓂᒌᔮᓂᖃᑦᑕᖅᓯᒪᔪᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ ᐱᕙᓪᓕᐊᓪᓚᑎᖕᓂᖏᑦ ᒪᓕᒃᑐᑦ ᐅᓄᖅᓯᒋᐊᖅᑕᐅᕙᓪᓕᐊᓂᖏᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ, ᐊᖑᓂᒃᑕᐅᔪᑦ ᑎᑭᐅᑎᓯᒪᓪᓗᑎᒃ ᖁᕝᕙᓯᑖᒥᒃ 2022–ᖑᑎᓪᓗᒍ. ᐊᖑᓂᒃᑕᐅᔪᑦ ᐅᓄᕈᓐᓃᖅᓯᒪᔪᑦ 32 990 ᑕᓐᓯᓄᑦ ᑭᖑᓂᐊᒍᑦ ᐅᓄᕈᓐᓃᕆᐊᖅᑎᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ 2023–ᒥ.

	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
ᑲᑎᓗᒃᑖᖅᓗᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ	30	30	32.3	32.3	36.4	36.4	36.4	36.4	33.3	33.3
ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 0	15.4	14.1	15.9	16.0	18.3	17.9	19.12	18.3	16.4	
ᐊᒡᒍᑐᖅᓯᒪᔪᖅ 1	14.9	15.2	16.2	16.2	18.0	18.1	17.3	18.8	16.6	
ᑲᑎᖢᒋᑦ STACFIS[1]	30.3	29.3	32.1	32.2	36.3	36.0	36.4	37.2	33.0	

[1] ᑐᙵᕕᒋᓪᓗᒋᑦ STATLANT, ᑐᓴᐅᒪᔾᔪᑎᖃᖅᖢᑎᒃ ᑲᓇᑕᒥ ᐊᒻᒪᓗ ᑲᓛᓖᓂᑦ ᐱᔭᓐᓇᕐᕕᖃᖅᑐᓂ ᐱᖃᓯᐅᔾᔨᙱᓪᓗᑎᒃ ᓯᒡᔭᖏᓐᓂ ᐊᖑᓂᒃᑕᐅᔪᑦ.

[2] STACFIS ᒥᒃᓴᐅᓴᒃᑕᐅᔪᑦ ᐊᑐᖅᖢᑎᒃ 1.48 ᐊᓯᙳᖅᑎᑦᑎᓂᕐᒧᑦ ᐱᔾᔪᑕᐅᔪᑦ ᑕᒪᑐᒧᖓ J–cut, ᓂᐅᕿᑕᐅᕙᒃᑐᓂ.

[3] ᐱᔭᐅᓪᓗᑎᒃ ᐊᖑᓂᒃᑕᐅᓪᓚᑦᑖᖅᑐᓂ ᑲᓛᓖᑦ ᑎᑎᕋᕐᕕᐊᓂ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᓚᐃᓴᓐᓯᒃᑯᓐᓂ (GLFK) ᓲᖃᐃᒻᒪ STATLANT ᐊᑐᐃᓐᓇᐅᙱᑦᑐᑦ.

ᑐᑭᓯᒋᐊᕈᑎᖃᕐᕕᐅᔪᑦ

SCR Docs. 24/013, 019, 020, 021, 022; SCS Doc. 24/14.

ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᑦ (PA-WG) 13 ᒪᐃ 2024. 09:00 Hᐊᓚᐸᒃᔅ ᓯᕿᙳᔭᖕᒐᒍᑦ

ᖃᕋᑕᐅᔭᒃᑯᑦ ᑲᑎᒪᔾᔪᑎ

ᐃᔅᓯᕙᐅᑕᖅ: ᐳᕐᓴᓐᑐ ᒐᓐᔭᓕᔅ–ᑯᔅᑕᔅ

1. ᒪᑐᐃᖅᓯᓂᖅ.

ᑲᑎᒪᓂᖅ ᒪᑐᐃᖅᑕᐅᓚᐅᐳᖅ ᐃᒃᓯᕙᐅᑕᕐᒧᑦ, ᐳᕐᓴᓐᑐ ᒐᓐᔭᓕᔅ–ᑯᔅᑕᔅᒧᑦ (ᔪᐊᕋᑉ ᑲᑐᔾᔨᖃᑎᒋᓂᒃ), ᐅᕗᖓᖅᑎᓪᓗᒍ 09:00 ᐃᑲᕐᕋᐃᑦ (ᐊᐅᓚᑕᐅᔪᖅ ᓯᓚᕐᔪᐊᕐᒥ ᓯᕿᙳᔭᖅ/ᑭᕙᓪᓕᐅᑉ ᓯᕿᙳᔭᖕᒐᒍᑦ ᑭᖑᕙᓯᓪᓗᓂ 3–ᓂᒃ ᐃᑲᕐᕋᓂᒃ Hᓚᐸᒃᔅ, ᓄᕙ ᓯᑰᔅ) ᓇᒡᒐᔾᔭᐅ, 13 ᒪᐃ 2024.

ᐃᒃᓯᕙᐅᑕᖅ ᑐᙵᓱᒃᑎᑦᑎᕗᖅ ᑭᒡᒐᖅᑐᐃᔪᓂᒃ ᑲᓇᑕᒦᖓᖅᓯᒪᔪᓂᒃ, ᔪᐊᕋᑉ ᑲᑐᔾᔨᖃᑎᒋᓂᒃ, ᔭᐹᓐᒥᑦ, ᐅᓚᓯᐊᑯᓂᑦ ᑲᑐᔾᔨᖃᑎᒋᓂᑦ, ᐊᒻᒪᓗ ᐊᒥᐊᓕᑲᐃᑦ ᓄᓇᖓᓐᓂᑦ, ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᖃᐃᖁᔭᐅᓯᒪᔪᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᓚᕆᑦ ᐱᓂᕐᓗᒃᑕᐅᔾᔭᐃᖅᓯᒪᓂᕐᒥᒃ ᐱᓕᕆᒋᐊᕐᓂᕐᒧᑦ ᐋᖅᑭᒃᓱᖅᓯᒪᕕᖓᓂ ᐃᖃᓗᒐᓯᐊᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒥ. ᓈᕆᔭᖅ ᐱᓕᕆᖃᑕᐅᔪᓂᒃ ᑎᑎᖅᓯᒪᔪᑦ ᐱᔭᐅᑎᑕᐅᕗᑦ ᓇᓂᑦᑎᓯ I–ᒥ.

a) ᑎᒃᑯᐊᖅᑕᐅᔪᖅ ᐅᓂᒃᑳᕆᐊᖅᑐᖅᑎᕈᑦᑎᑕᐅᓪᓗᓂ.

ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐊᐅᓚᑦᑎᕙᖓᑦ (ᑕᐃᓇ ᕙᐊᓪ ᒪᒃᑲᓪᓚᒻ ᐊᒻᒪᓗ ᔭᓴ ᐊᑯᕐ) ᓂᕈᐊᕐᓴᒃᓴᓕᐊᕆᔭᐅᓚᐅᖅᐳᒃ ᐅᓂᑳᕆᐊᖅᑐᖅᑎᐅᖁᔭᐅᓪᓗᓂ ᑲᑎᒪᓂᕆᔭᐅᔪᒥᒃ.

b) ᓈᒻᒪᒋᔭᐅᓂᖓ ᑲᑎᒪᔾᔪᑎᓴᐅᑉ

ᑲᑎᒪᔾᔪᑎᒃᓴᑦ ᓈᒻᒪᒋᔭᐅᕗᖅ ᑕᐃᒪᓐᓇ ᑐᓂᐅᖃᑕᐅᓂᖏᓐᓄᑦ (ᓇᓂᑦᑎᓯᖓ II).

2. ᐱᓕᕆᓂᕆᔭᐅᔪᖅ ᓈᓴᐅᓯᕆᓂᕐᓂᒃ ᐊᐅᓚᑦᑎᓂᒻᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᓂᑦ

ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᑦ (PA-WG) ᕿᒥᕐᕈᓚᐅᕐᒪᑕ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᓂᒃ ᑕᐃᒪᓐᓇ ᐊᖏᖅᑕᐅᖕᒪᑕ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᓂᒃ–ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐱᓕᕆᒋᐊᕈᑎᒃᓴᐃᑦ (WG-RBMS) ᐊᐃᕆᓕ 2024 ᑲᑎᒪᓂᕐᒥ (COM SC ᑎᑎᕋᒃᓯᒪᔪᑦ. 24-01). ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᖁᓪᓗᒋᑦ ᑕᒪᒃᑯᐊ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᓴᖅᑭᑎᑕᐅᓚᐅᖅᐳᑦ ᓚᕿᐊᓄ ᑯᐅᓐ–ᐊᓗᓐᓴᒧᑦ ᑭᒡᒐᖅᑐᖅᑐᒍ ᐋᖅᑭᑦᑎᐊᖅᓯᒪᔭᕆᐊᓕᖕᓂᑦ ᐱᓕᕆᖅᑲᑎᒌᒃᑐᓂᒃ ᐊᒻᒪᓗ ᐅᖃᖅᑲᑎᒌᒍᑎᒋᔭᐅᓪᓗᑎᒃ ᑲᑎᑕᐅᓯᒪᔪᓄᑦ. ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᑦ (PA-WG) ᓈᒻᒪᒃᓴᓚᐅᖅᐳᑦ ᐱᓕᕆᓃᑦ ᓈᓴᐅᓯᕆᓂᕐᒥᒃ ᖃᐅᔨᓴᖅᑕᐅᖁᓪᓗᒍ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᑎᑎᕋᒃᑕᐅᓯᒪᔪᓂᑦ ᑎᑎᕋᒃᓯᒪᔪᖅ 1–ᒥ, ᐅᖃᐅᓯᖃᒃᑐᓂᒃ ᑕᒪᒃᑯᐊ ᐱᓕᕆᒋᐊᓕᓂᐅᓂᖏᓐᓂᒃ ᐱᓕᕆᐊᕆᐅᖁᓪᓗᒋᑦ ᖃᐅᔨᓇᓱᐋᖅᓯᓂᑦ, ᑭᓯᐊᓂ ᐱᔭᕇᒃᓯᔾᔪᑎ ᐋᖅᑭᒃᓱᖅᓯᒪᕕᖓᓂᑦ ᐊᒻᒪᓗ ᖁᒃᓴᓇᕐᓂᖓᑕ ᖃᓄᑎᒋᐅᓂᖓ ᐋᖅᑭᑕᐅᔭᕆᐊᖃᕐᓂᐊᕐᒪᑦ ᐃᓱᒪᓕᐅᕈᑎᒋᔭᐅᓗᓂᓗ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᓂᒃ ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐱᓕᕆᒋᐊᕈᑎᒃᓴᐃᑦ (WG-RBMS) ᑲᑎᒪᓂᕐᒥ ᐋᒡᒋᓯ 2024–ᒥ.

ᓇᓗᓇᐃᒃᑯᑎ 1. ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐊᒻᒪᓗ ᑐᕌᖓᔪᑦ ᐱᓕᕆᓃᑦ ᓈᓴᐅᓯᕆᓂᕐᒥᒃ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᓂᕐᓗᒃᑕᐅᔾᔭᐃᖅᓯᒪᓂᕐᒥᑦ ᐱᓕᕆᐊᕐᓂᑦ ᐋᖅᑭᒃᓱᖅᓯᒪᕙᖕᓂᒃ. ᐃᓚᖏᔭᐅᓗᑎᒃ ᐱᓕᕆᔾᔪᓯᕐᒧᑦ ᓇᓗᓇᐃᑯᑕᕐᓄᑦ ᓱᕐᓗ B ᐊᒻᒪᓗ F ᓇᓗᓇᐃᖅᑕᐅᕐᑯᓪᓗᒋᑦ ᐱᕙᐅᔪᓐᓇᖅᑐᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪᓗ ᐃᖃᓗᖓᓱᐊᕐᓂᑦ ᑐᕐᑯᓂᓇᖁᒃᑕᖏᑦ ᑭᒡᖓᖅᑐᖅᑕᐅᓗᑎᒃ, ᐊᒻᒪᓗ ᓇᓂᑦᑎᓯᒋᓐᓅᖓᔪᑦ ᐊᒻᒪᓗ/ᐅᖁᔫᓐᓃᑦ ᐋᑑᑎᓪᖏᓄᑦ ᓇᓗᓇᐃᖅᓯᔾᔪᑎᖏᓐᓄᑦ ᓲᕐᓗ B_{lim}, $B_{trigger}$, B_{msy}, F_{lim}, F_{msy}, ᐊᒻᒪᓗ MSY, ᐃᓚᖏᑦ ᑕᒪᐃᑎᒍᑦ ᑎᑎᕋᒃᓯᒪᔪᑦ ᐊᑐᖅᑕᐅᔪᑦ ᐃᓅᓇᓕᒫᖓᓂ ᑖᔅᓯᒪ ᑎᑎᕋᓯᒪᔮ ᐃᓚᖏᔭᐅᓪᓗᓂ t ᓇᓗᓇᐃᖅᑕᐅᕐᑯᓪᓗᒍ ᐊᕐᕌᖓ, t=1 ᐅᖁᔫᓐᓃᑦ t_1 ᓇᓗᓇᐃᖅᑕᐅᕐᑯᓪᓗᒍ ᓯᕗᓪᓕᖅ ᐊᕐᕋᒍᖓ ᑕᐃᔅᓯᒪᓂᓪᓚᓇᒃ ᓴᖅᑭᑎᑕᐅᕕᒋᔭᖓᓂ ᑭᖑᓂᖓᒍᑦ ᓄᖑᕝᕕᒋᔭᖓᑕ, t=x ᐅᖁᔫᓐᓃᑦ t_x ᓇᓗᓇᐃᖅᑕᐅᕐᑯᓪᓗᒍ ᐊᕐᕋᒍᖓ x ᓴᖅᑭᑎᑕᐅᓂᖓᑕ, T ᓇᓗᓇᐃᖅᑕᐅᕐᑯᓪᓗᒍ ᐃᓱᓕᔾᔪᑎᖓ ᐊᕐᕋᒍᑉ ᓴᖅᑭᑎᑕᐅᓂᖏᓐᓂ, ᑕᒪᓐᓇ ᓇᓗᓇᐃᖅᑕᐅᓰᖑᓪᓗᓂ ᐃᒪᓐᓇᐅᓪᓗᒍ 50 ᐊᕐᕋᒍᐃᑦ ᑭᓯᕋᔭᐅᓂᖏᓐᓂᑦ ᖃᐅᔾᔭᖅᓯᔾᔪᑎᒋᔭᐅᓪᓗᓂ ᐊᒻᒪᓗ 25 ᐊᕐᕋᒍᓂᑦ ᑕᐃᒃᑯᓂᖓᓪᓗᐊᖅ ᖃᐅᔨᓇᓱᐊᖅᓯᓂᐅᔪᓂᑦ, ᐊᒻᒪᓗ t_{Btr} ᓇᓗᓇᐃᖅᑕᐅᕐᑯᓪᓗᒍ ᐊᕐᕋᒍᖓ ᑕᐃᑦᓯᒪᓂ ᐊᒥᓲᓂᖏᑦ ᑎᑭᐅᑎᑎᓪᓗᒋᑦ $B_{trigger}$ $\left(B_{t_{Btr}} \geq B_{trigger}\right)$ ᑭᖑᓂᖓᒍᑦ ᐱᒋᐊᓚᐅᑦᑎᓪᓗᒋᑦ ᓄᖑᑕᐅᓯᒪᓂᖏᓐᓂᑦ ᐱᕙᒃᓴᐅᓂᓇᓚᐅᖅᑕᕐᒥᓂᑦ. ᐊᖑᓇᓱᐊᕐᓂᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒪᓕᒃᑕᐅᔭᕆᐊᓕᑦ (HCR) ᐊᑐᖅᑕᐅᓰᖑᒪᑦ ᓇᓗᓇᐃᖅᓯᔾᔪᑎᒋᔭᐅᓪᓗᓂᐊᖑᓇᓱᐊᕐᓂᕐᒥᑦ ᐅᑯᓇᓂ ᐊᖑᓇᓱᐊᕈᓐᓇᖅᑎᑕᐅᓂᕐᒧᑦ ᐱᓕᕆᐊᕈᑎᒃᓴᒥ ᑐᒃᒥᖒᓇᔭᐅᓯᒪᔪᒥᑦ ᐱᓂᕐᓗᒃᑕᐅᔾᔭᐃᖅᓯᒪᓂᕐᒥ ᐱᓕᕆᐊᕐᓂᖅ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ (PA) ᐋᖅᑭᒃᓱᖅᓯᒪᕙᖕᓂ, ᐊᒻᒪᓗ F=0 ᐊᑐᖅᑕᐅᓰᖑᒪᑦ ᓇᓗᓇᐃᖅᓯᔾᔪᑎᒋᔭᐅᓪᓗᓂ ᐊᖑᔭᖅᑲᖏᓐᓂᖓᓂᒃ.

ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔭᖅ	ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᖁᖖᒐᑦ ᖁᒃᓴᓇᖅᑐᓕᓂᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐱᓕᕆᐊᕈᑎᒃᓴᐃᑦ (RBMS) ᐅᓂᒃᑳᖅᓯᒪᔪᒥᑦ	ᐊᐅᓚᔪᑎᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᑯᓇᖖᒐᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐱᓂᕐᓗᒃᑕᐃᔾᔭᐃᖅᓯᒪᓂᕐᒥᑦ ᐱᓕᕆᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓴᐃᔾᔭᖅᑎᒋᒃᑐᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᒋᒃᑐᓄᑦ (SC PAWG)	ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᓂᑦ	ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ
ᐱᓂᕐᓗᒃᑕᐅᔭᐃᖅᓯᓯᒪᓂᖅ ᐱᐅᖏᑦᑐᓂᑦ ᐱᕙᒃᓴᐅᔪᓂᑦ ᐅᖃᒃᓯᒪᖕᒪᑕ ᓴᖅᑭᓐᓂᖏᓐᓂᒃ ᐅᐋᑦᑎᐊᕈᓪᓚᖅᑎᓪᓗᒍ ᐱᕙᒃᓴᐅᔪᑦ ᖃᓄᐃᕐᓗᒍᑎᖃᑦᑎᓐᓇᒋᑦ ᐊᕙᒃᓯᒪᓂᕐᒥ. (ᖃᐅᔾᓴᖅᓯᓂᑦ ᐱᒋᐊᕙᖃᕐᒪᑕ ᒪᑯᓂᖓ ᓄᖑᑉᐸᓪᓕᐊᖏᑦᑐᓂᒃ ᐱᕙᒃᓴᐅᔪᓂᑦ)	ᖁᒃᓴᓇᓗᐊᕌᓗᒃᑎᓇᒋᑦ ᐱᕙᒃᓴᐅᔪᑦ ᓄᖑᓕᕐᓂᖏᓐᓂᑦ	ᖁᒃᓴᓇᓗᐊᕌᓗᒃᑎᓇᒋᑦ ᐱᕙᒃᓴᐅᔪᑦ ᓄᖑᓕᕐᓂᖏᓐᓂᑦ	$P(B_t < B_{lim}) \leq 0.10$	ᕿᑎᖓᓃᑦᑐᑦ ᐃᓚᖓᓂ ᐊᕐᕋᒍᐃᑦ ᓴᖅᑭᑎᑕᐅᓂᖓᓂ (t=1 ᑎᑭᓪᓗᒍ T) ᑕᐃᒃᑯᓇᓂ ᐱᕙᒃᓴᐅᔪᑦ ᑐᖖᒑᓂᑎᓪᓗᒋᑦ B_{lim}. ᐊᒃᐸᓯᒃᑐᒪᓇᐅᑎᓪᓗᒍ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖅᑲᓂᖏᑦ ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᖕᒪᑦ ᐃᒪᓐᓇᐅᓗᓂ 10%−ᖓ ᑐᖖᒐᕙᒋᓗᒍ ᒫᓐᓇ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ (PA). ᐃᓚᒋᔭᐅᓗᓂ ᐅᒨᖓ PS, ᓯᐊᒻᒪᖅᑎᑕᐅᓂᖏᑦ ᑕᒪᒃᑯᐊ ᐊᑐᑐᐃᓐᓇᓈᓕᑦ, ᐃᓚᖏᑦ ᐱᓕᕆᐊᕆᔭᐅᓪᓗᐊᑕᖅᑐᑦ, ᑐᓴᖅᑎᑦᑎᔾᔪᑎᒋᔭᐅᔭᓇᐊᖅᑲᖅᒪᑕ ᖃᓄᐃᑦᑐᓂᖏᓐᓂᒃ ᐅᖁᔫᓐᓃᑦ ᓯᐊᒻᒪᖅᑎᑕᐅᓂᖏᓐᓂᒃ (ᑐᑭᓕᐅᑎᓗᒍ ᐊᑦᑕᖅᑐᔪᖅ ᐊᑭᓴᖅᑐᖅᓗᒍ ᑐᐊᑦᑐᖅ).

ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ	ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᕙᓐᖓᑦ ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᕐᑎᓂᕐᒥᑦ ᐱᓕᕆᒥᐊᕈᑎᒃᓴᐃᑦ (RBMS) ᐅᓂᒃᑳᖅᓯᒪᔪᒥᑦ	ᐊᐅᓚᔪᑎᓂᒃ ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᑯᓇᓐᖓᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐱᓂᕐᓗᒃᑕᐃᔾᔭᐃᖅᓯᒪᓂᕐᒥᑦ ᐱᓕᕆᒥᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓴᐃᔭᖅᑲᑎᒋᒃᑐᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᓄᑦ (SC PAWG)	ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᓂᑦ	ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ
		ᖁᒃᓴᓇᕐᓂᕐᒥᑦ ᐱᕙᒃᓴᐅᔪᑦ ᑐᓐᓛᓄᓕᖅᑎᓪᓗᒥᑦ $B_{trigger}$	$P(B_t < B_{trigger}) \le 0.30$	ᕿᑎᖓᓃᑦᑐᖅ ᐃᓚᖓᑕ ᐊᕐᕌᒍᐃᑦ ᓴᖅᑭᑎᑕᐅᕙᖕᒥᑕ (t=1 ᑎᑭᓪᓗᖕᒍ T) ᑕᐃᔅᓱᒪᓂ ᐱᕙᒃᓴᐅᔪᑦ ᑐᓐᓛᓂᓕᖅᑎᓪᓗᒥᑦ $B_{trigger}$. ᐊᒃᐸᓯᒃᑎᓪᓗᒍ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖃᓂᖓ ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᖕᒪᑦ ᐃᒪᓐᓇᐅᓗᓂ 30%−ᖓ. ᐱᖃᓯᐅᑎᓗᒥᑦ ᐱᔭᕇᒃᑕᐅᔾᔪᑎᖏᑦ ᐅᑯᐊ F=0 ᑐᓴᖅᑎᑦᑎᔾᔪᑎᒥᔭᐅᑦᑎᐊᕐᓂᖅᓴᐅᕐᑯᓪᓗᒍ ᐱᔭᕇᒃᑕᐅᔪᑎᖏᓐᓂᒃ ᐱᓕᕆᐊᕆᔭᐅᓚᓂᖏᑎᒍᑦ ᐅᑯᐊ ᐊᖑᓇᓱᐊᕐᓂᕐᒥᒃ ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᒪᓕᒃᑕᐅᔭᕆᐊᓕᑦ (HCR) ᐊᑭᓱᖅᑐᕐᓗᖕᒍ ᐊᔾᔨᒌᖕᒍᓂᓱᖕᒍᓂᖏᑦ ᐱᕙᒃᓴᐅᔪᑦ ᐊᒃᓱᕈᒃᑎᑦᑎᔪᓐᓇᕐᓂᖕᒥᑦ.
	ᐋᖅᑭᐅᒪᑎᓪᓗᒥᑦ ᐱᕙᒃᓴᐅᔪᑦ ᐅᖓᑖᓂᓪᓗᑎᒃ B_{msy} ᐅᓄᖅᑐᐃᖅᓯᕐᓂᖅᓴᐅᓗᑎᒃ ᑕᐃᒪᐃᖅᑕᖅᑎᖅᒥᓂᖅᓴᐅᓂᖏᓐᓂ	ᐋᖅᑭᐅᒪᑎᓪᓗᒥᑦ ᐱᕙᒃᓴᐅᔪᑦ ᐅᖓᑖᓂᓪᓗᑎᒃ B_{msy} ᐅᓄᖅᑐᐃᖅᓯᕐᓂᖅᓴᐅᓗᑎᒃ ᑕᐃᒪᐃᖅᑕᖅᑎᖅᒥᓂᖅᓴᐅᓂᖏᓐᓂ	$P(B_t/B_{msy} > 1) \le 0.30$	ᕿᑎᖓᓂ ᐃᓚᖏᑕ ᐊᕐᕌᒍᐃᑦ ᐅᑯᓇᓂ $\frac{B_t}{B_{msy}}$ ᐊᖏᓂᖅᓴᐅᑎᓪᓗᒍ ᐅᒪᓐᖓᑦ 1 ᓴᖅᑭᑕᐅᓯᒪᓂᖏᓐᓂ (t=1 ᑎᑭᓪᓗᒍ T). ᑕᐃᒪᓐᖓᓂᑦ 50%−ᖏᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᔪᐊᓘᖅᑎᓪᓗᒋᑦ ᓈᔭᐅᖕᒪᑕ 51-99%−ᖏᑕ ᑎᑭᐅᒪᔭᖏᑦ, ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᓯᒪᕗᑦ ᑕᕝᕙᓂ ᕿᑎᖓᓂ ᑖᒃᑯᐊ ᑎᑭᐅᒪᔭᖏᑕ (75%) ᐱᔪᒥᓇᕆᔭᐅᔪᒥᑦ ᐊᖏᖃᑎᒌᒍᑎᖃᕈᓐᓇᖅᓯᓇᓱᐊᕐᓂᖏᓐᓂ ᐅᑯᓇᓂ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑲᑎᒋᒃᑐᓄᑦ (PAWG)/ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᕐᑎᓂᕐᒥᑦ ᐱᓕᕆᒥᐊᕈᑎᒃᓴᐃᑦ (RBMS).
	ᖁᒃᓴᓇᓗᐊᖏᑦᑐᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ ᐃᖃᓗᒃᑕᐅᓂᐊᓇᐊᖓᑕ	ᖁᒃᓴᓇᓗᐊᖏᑦᑐᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ ᐃᖃᓗᒃᑕᐅᓂᐊᓇᐊᖓᑕ	$P(F_t/F_{msy} < 1) \le 0.70$	ᕿᑎᖓᓂ ᐃᓚᖓᑕ ᐊᕐᕌᒍᐃᑦ ᐅᑯᓇᓂ $\frac{F_t}{F_{msy}}$ ᐃᒪᓐᓇᐅᑎᓪᓗᒥᑦ ᑐᓐᓛᓂᑎᓪᓗᒥᑦ 1 ᐅᑯᓇᓂ ᓴᖅᑭᑕᐅᓂᖏᓐᓂ (t=1 ᑎᑭᓪᓗᖕᒍ T). ᐊᒃᐸᓯᒃᑐᖅ ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐊᖃᖅᓂᖓ ᐊᐅᓚᔾᔪᑎᖃᓕᖅᑎᑕᐅᑎᓪᓗᒍ ᐃᒪᓐᓇᑎᒥᓂᒃ 30%−ᖏᓐᓂᒃ ᑐᓐᖓᕕᒥᓗᒍ ᒫᓐᓇᕋᑖᖑᓚᐅᖅᑐᖅ ᐅᐊᓐᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐊᐅᓚᕐᑎᓂᖅ ᐸᕐᓇᒍᑎᓕᐊᕆᓯᒪᔪᒥᒃ ᖃᐅᔨᓴᖅᓯᓂᕐᒧᑦ (MSE).

ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ	ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᕙᙵᑦ ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐱᓕᕆᐊᕈᑎᒃᓴᐃᑦ (RBMS) ᐅᓂᒃᑳᖅᓯᒪᔪᒥᑦ	ᐊᐅᓚᔪᑎᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᑯᓇᙵᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐱᓂᕐᓗᒃᑕᐃᔾᔭᐃᖅᓯᒪᓂᕐᒥᑦ ᐱᓕᕆᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓴᐃᔭᕐᑲᑎᒋᒃᑐᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᕐᑲᑎᒋᒃᑐᓄᑦ (SC PAWG)	ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᓂᑦ	ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ
ᖃᓄᐃᑦᑐᓐᓃᕐᓂᕐᒥᑦ ᐱᐅᔪᑦ ᐱᕕᒃᓴᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᕐᒥᑦ ᑕᐃᑦᓱᒪᓂ ᐱᕕᒃᓴᐅᔪᑦ ᐃᒃᐱᒋᔭᐅᑦᑎᐊᕆᐊᖃᒃᑎᑦᑐᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᔾᔨᖅᑐᖅᑕᐅᑦᑎᐊᕆᐊᖃᒃᑎᑦᑐᒥᑦ ᐊᕕᒃᓯᒪᓂᕐᒥ. (ᖃᐅᔨᓇᓱᐊᖅᓯᓂᖅ ᐱᒋᐊᓱᖑᖕᒪᑦ ᓄᖑᑕᐅᓯᒪᔪᓂᑦ ᐱᕕᒃᓴᐅᔪᓂᑦ)	ᐋᖅᑭᒃᓱᒃᑲᓐᓂᕐᓗᒥᑦ ᐱᕕᒃᓴᐅᔪᑦ ᐅᑯᐊ B_{msy}	ᐋᖅᑭᒃᓱᒃᑲᓐᓂᕐᓗᒥᑦ ᐱᕕᒃᓴᐅᔪᑦ ᐊᕙᑎᖏᓐᓄᑦ ᐅᑯᐊ B_{msy}	$P\big(\bar{\mu}(B_{T-10:T}) > B_{trigger}\big) \geq 0.80$	ᐃᓚᖓ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᑕ ᐊᑐᐊᖅᓯᖕᒪᑦ ᐊᒥᓲᓂᓪᓗᐊᑕᓇᓲᖏᓐᓂ ($\bar{\mu}$) ᐊᓂᒍᖅᑐᓂ 10 ᐊᕐᕌᖑᓂ ᓴᖅᑭᑕᐅᓯᒪᓂᖏᑕ ($T-10$ ᑎᑭᓪᓗᒍ T) ᐊᖏᓂᖅᓴᐃᖕᒪᑦ ᐅᒪᖓᑦ $B_{trigger}$. ᑕᐃᒪᐃᑐᐃᓐᓇᓇᐋᖅᑲᓗᐊᖏᑦᑐᖅ [ᐋᖅᑭᒃᓱᒃᑲᓂᖅᑎᓐᓇᒍ ᐊᕙᑎᖏᓐᓄᑦ ᐅᑯᐊ B_{msy}] ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᖕᒪᑕ ᐃᒪᓐᓇᐅᓗᒋᑦ 20% ᑐᙵᕕᒋᓗᒍ ᒫᓐᓇ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ (PA).
	ᐱᐅᓈᖅᑐᑦ ᐱᕕᒃᓴᐅᔪᑦ ᖃᓄᐃᑦᑐᓂᖅᓂᕐᖓᓂᑦ ᐱᓕᕆᓂᖅ	ᖃᐅᔨᒪᒋᐊᕐᓗᒍ ᕿᓚᒻᒥᐅᔪᖅ ᐊᑐᕐᓂᖓᓂ ᐱᕈᒃᐸᓪᓕᐊᓂᖅ	$P(B_{t=5} > B_{t=1}) \geq 0.75$	ᐃᓚᖓ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᑕ ᐊᑐᐊᖅᓯᖕᒪᑦ ᐅᑯᓇᓂ $B_{t=5} > B_{t=1}$ ᑕᐃᒪᙵᓂᑦ 50%−ᖏᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᔪᓐᓃᖅᑎᓪᓗᒋᑦ, ᐃᓱᒪᒋᔭᐅᓪᓗᓂ “ᑕᐃᒪᐃᒐᔪᖕᓂᖅᓴᖅ ᑕᐃᒪᐃᖅᑲᑦᑕᖏᓐᓂᖓᓂᓪᓚ” ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᓯᒪᖕᒪᑦ ᐃᒪᓐᓇᑎᒋᐅᓗᓂ 75%.
		ᖃᐅᔨᒪᒋᐊᕐᓗᒍ ᕿᑎᖓᓂ ᐊᑐᕐᓂᐊᖅᑐᖅ ᐱᕈᕐᓂᖅ	$P(B_{t=15} > B_{t=1}) \geq 0.75$	ᐃᓚᖓ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᑕ ᐊᑐᐊᖅᓯᖕᒪᑦ ᐅᑯᓇᓂ $B_{t=15} > B_{t=1}$ ᑕᐃᒪᙵᓂᑦ 50%−ᖏᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᔪᓐᓃᖅᑎᓪᓗᒋᑦ, ᐃᓱᒪᒋᔭᐅᓪᓗᓂ “ᑕᐃᒪᐃᒐᔪᖕᓂᖅᓴᖅ ᑕᐃᒪᐃᖅᑲᑦᑕᖏᓐᓂᖓᓂᓪᓚ” ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᓯᒪᖕᒪᑦ ᐃᒪᓐᓇᑎᒋᐅᓗᓂ 75%.
		ᖃᐅᔨᒪᒋᐊᕐᓗᒋᑦ ᐊᑯᓂ ᐊᑐᕐᓂᐊᖅᑐᖅ ᐱᕈᕐᓂᖅ	$P(B_{t=25} > B_{t=1}) \geq 0.75$	ᐃᓚᖓ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᑕ ᐊᑐᐊᖅᓯᖕᒪᑦ ᐅᑯᓇᓂ $B_{t=25} > B_{t=1}$ ᑕᐃᒪᙵᓂᑦ 50%−ᖏᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᔪᓐᓃᖅᑎᓪᓗᒋᑦ, ᐃᓱᒪᒋᔭᐅᓪᓗᓂ “ᑕᐃᒪᐃᒐᔪᖕᓂᖅᓴᖅ ᑕᐃᒪᐃᖅᑲᑦᑕᖏᓐᓂᖓᓂᓪᓚ” ᐊᐅᓚᔪᓐᓇᖅᓯᑎᑕᐅᓯᒪᖕᒪᑦ ᐃᒪᓐᓇᑎᒋᐅᓗᓂ 75%.
		ᐱᕕᖃᕐᓂᖅ ᑕᓯᕋᓯᐊᕐᓗᒋᑦ (ᐃᓘᓐᓇᑕᒫᖏᑦ)	$Count(t_1: t_{Btr})$	ᕿᑎᖓᓃᑦᑐᑦ ᐊᒥᓲᓂᖏᑦ ᐊᕐᕌᒍᐃᑦ (t_{Btr}) ᑎᑭᑕᐅᓇᓱᐊᕐᓗᓂ $B_{trigger}$ ᑖᓐᓇ ᒥᐊᑐᓇᒃ ᐋᖅᑭᕕᒃᓯᒪᔪᒥᑦ ᑲᔪᓯᔪᑎᖅᑲᕐᒥᖕᒪᑦ ᖃᐅᔨᓴᖅᓯᕐᔪᑎᒋᔭᐅᓯᖅ ᒪᓕᑐᐃᓐᓇᕐᓂᖓᓄᑦ ᑕᐃᒃᑯᓂᖓᓪᓗᐊ ᐱᕕᐅᔪᓐᓇᖅᑐᓂᒃ ᐊᑐᖅᓯᒪᔭᖏᓐᓂ.

ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ	ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᕙᙵᑦ ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐱᓕᕆᒋᐊᕈᑎᒃᓴᐃᑦ (RBMS) ᐅᓂᒃᑳᖅᓯᒪᔪᒥᑦ	ᐊᐅᓚᔪᑎᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᑯᓇᙵᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐱᓂᕐᓗᒃᑕᐃᔾᔭᐃᖅᓯᒪᓂᕐᒥᑦ ᐱᓕᕆᒋᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓴᐃᔭᖅᑎᒋᒃᑐᑦ ᖃᐅᔨᓇᓱᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᒃᑯᓐᓄᑦ (SC PAWG)	ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᓂᑦ	ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ
		ᐱᕕᖃᖅᓂᖅ ᑕᓯᕋᓯᐊᕐᓗᒋᑦ (ᑕᒪᒃᑯᓄᙵᖓᔪᑦ)	$Ratio\left(\frac{Count^{F=HCR}(t_1:t_{Btr})}{Count^{F=0}(t_1:t_{Btr})}\right) \leq 12$	ᕿᑎᖓᓃᑦᑐᑦ ᐅᓄᓚᓈᖕᓂᖏᓐᓂᑦ ᐊᑯᓐᓂᖏᓐᓂ ᐊᒥᓲᓂᖏᑕ ᐊᕐᕌᒍᐃᑦ (t_{Btr}) ᑎᑭᑕᐅᓇᓱᐊᕐᓗᓂ $B_{trigger}$ ᐅᑯᓇᓃᑦᑐᑦ HCR (ᐊᖑᓇᓱᐊᕐᓂᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒪᓕᒃᑕᐅᔭᕆᐊᓕᑦ) ᐊᑭᓴᖅᑐᕐᓗᒍ ᐅᑯᑎᒎᓇ F=0. ᑖᓐᓇ ᒥᐊᑐᓇᒃ ᐋᖅᑭᕕᒃᓯᒪᔪᒥᑦ ᑲᔪᓯᔾᔪᑎᖅᑲᖏᒻᒪᑦ ᖃᐅᔨᓴᖅᓯᔾᔪᑎᒋᔭᐅᓯᖅ; ᐃᓯᒪᖅᔩᑎᒋᔭᐅᓪᓗᓂ ᐋᖅᑭᒃᑕᐅᓯᒪᖕᒪᑦ ᑕᕝᕙᓂ ᐃᒪᓐᓇᑎᒋᐅᓪᓗᒍ 20% ᐅᕝᕙᓘᓐᓃᑦ ᐅᓄᖏᓐᓂᖅᓴᐅᓗᓂ F=0 ᐱᔪᒥᓇᓇᓴᐅᔪᒥᑦ ᐊᖏᖅᑲᑎᒌᒍᑎᒋᔭᐅᓇᓱᐋᖅᓂᖓᓂ ᐅᑯᓇᙵᑦ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᒃᑯᓐᓄᑦ (PAWG)/ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᐱᓕᕆᒋᐊᕈᑎᒃᓴᐃᑦ (RBMS)
		ᐱᕕᖃᖅᓂᖅ ᑕᓯᕋᓯᐊᕐᓗᒋᑦ (ᐃᓚᒃᑲᓐᓂᖏᑦ ᐊᕐᕌᒍᐃᑦ)	$t_{extra} = Count^{F=HCR}(t_1:t_{Btr}) - Count^{F=0}(t_1:t_{Btr})$	ᕿᑎᖓᓃᑦᑐᑦ ᓈᓴᐅᑎᑦ ᐃᓚᒃᑲᓐᓂᖏᑦ ᐊᕐᕌᒍᐃᑦ (t_{extra}) ᑎᑭᑕᐅᓇᓱᐊᕐᓗᓂ $B_{trigger}$ ᐅᑯᓇᓃᑦᑐᑦ HCR (ᐊᖑᓇᓱᐊᕐᓂᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒪᓕᒃᑕᐅᔭᕆᐊᓕᑦ) ᐊᑭᓴᖅᑐᕐᓗᒍ ᐅᑯᑎᒎᓇ F=0. ᑖᓇ ᒥᑐᓇᒃ ᐋᖅᑭᕕᒃᓯᒪᔪᒥᒃ ᑲᔪᓯᑦᑎᐊᖅᑐᒥᒃ ᐋᖅᑭᒃᓯᐃᔪᑎᖅᑲᖏᒻᒪᑦ.
ᐊᒥᒐᖅᒋᑦᑐᒥᒃ ᓈᒻᒪᒋᔭᐅᔪᖅ ᐃᖅᑲᓗᒐᓱᐊᓂᕐᒧᑦ ᐱᓕᕆᓂᐅᔪᖅ ᐃᓘᓐᓇᓕᒫᖓᓂ ᐱᕕᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐅᖅᑲᐅᓯᒪᖕᒪᑦ ᐃᓗᐊᓂ ᐅᔾᔨᕐᓱᖅᑎᐊᓂᕐᒥ ᐊᒻᒪᓗ ᖃᓄᐃᖅᓱᒍᑎᖅᑲᕐᑐᓂ ᐊᕙᒃᑐᖅᓯᒪᓂᖅᓂ. (ᖃᐅᔨᓇᓱᐊᖅᓯᓂᖅ ᐱᒋᐊᓱᖑᒪᑦ ᓄᖑᑕᐅᓯᒪᔪᓂᑦ ᐱᕕᒃᓴᐅᔪᓂᑦ)	ᐋᖅᑭᐅᒪᑎᑦᑎᓗᓂ ᒥᒃᓵᓂᑦᑐᓂᒃ MSY ᐱᔭᐅᔪᓂ ᐊᑯᓂ–ᑕᐅᕗᖓᑐᖅᑲᒃ	ᐋᖅᑭᐅᒪᑎᑦᑎᓗᓂ ᒥᒃᓵᓂᑦᑐᓂᒃ MSY ᐱᔭᐅᔪᓂ ᐊᑯᓂ–ᑕᐅᕗᖓᑐᖅᑲᒃ	$P\left(0.8 \geq \frac{Median(C_{T-10:T})}{MSY} < 1.2\right) \geq 0.80$	ᐃᓚᖓ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖓᑕ ᐊᑐᐊᔅᓯᖕᒪᑦ ᑕᒪᒃᑯᐊ ᐅᓄᓕᓈᓂᖏᑕ ᐊᑯᓐᓂᖏᓐᓂ ᕿᑎᖏᓐᓂ ᐱᔭᐅᔪᑦ ᐊᑐᓚᐅᑦᑐᓂ 10–ᓂ ᐊᕐᕌᒍᓂ ᓴᖅᑭᑕᐅᓂᖏᓐᓂ ᐊᒻᒪᓗ MSY ᐃᓗᐋᓂᒻᒪᑦ 0.8 MSY-1.2 MSY ᑎᑭᐅᒪᔭᖓᓂ. ᐊᑉᐸᓯᑦᑐᖅ ᑕᐃᒪᓐᓇᐅᑐᐃᓐᓇᓇᐊᖅᑲᓂᖏᓐᓂ [ᐋᖅᑭᐅᒪᑎᑕᐅᖏᓐᓂᖏᓐᓂ ᐱᔭᐅᔪᑦ ᒥᒃᓵᓂᑦᑐᑦ MSY] ᐊᐅᓚᔾᔪᑎᓕᐊᕆᔭᐅᓯᒪᖕᒪᑦ ᐃᒪᓐᓇᐅᓪᓗᒍ 20% ᑐᙵᕕᖅᑲᓪᓗᓂ ᒫᓐᓇ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑲᑎᒌᖏᑦ (NAFO) ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ (PA).

ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ	ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᕙᙵᑦ ᖁᒃᓴᓇᖅᑐᓕᕆᓂᕐᒥᑦ ᐊᐅᓚᕐᑎᓂᕐᒥᑦ ᐱᓕᕆᒥᐊᕈᑎᒃᓴᐃᑦ (RBMS) ᐅᓂᒃᑳᖅᓯᒪᔪᒥᑦ	ᐊᐅᓚᔪᑎᓂᒃ ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕆᔭᐅᔪᒪᔪᑦ ᐅᑯᓇᙵᑦ ᖃᐅᔨᓇᓯᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐱᓂᕐᓗᒃᑕᐃᔅᔭᐃᖅᓯᒪᓂᕐᒥᑦ ᐱᓕᕆᒥᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓴᐃᔭᖅᑎᒥᒃᑐᑦ ᖃᐅᔨᓇᓯᐊᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ ᐅᔾᔭᕐᓯᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓯᐊᖅᑐᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᒥᒃᑐᓄᑦ (SC PAWG)	ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᓂᑦ	ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ
	ᐃᖃᓗᒐᓱᐊᖅᑎᑦᑎᐊᕙᐃᑦ ᐱᓕᕆᑦᑎᐊᕐᓂᕐᒥᑦ	ᐅᒃᑐᕐᓗᒥᑦ ᐊᑯᓐᓂᖅᒥᓐᓂ–ᐊᕐᕌᒍᑕᒪᑦ ᐅᓄᓐᓂᖅᒥᑦ ᐱᔭᐅᔪᓐᓇᕐᑎᑕᐅᔪᑦ ᐊᔾᔨᒌᒍᓐᓃᕐᓂᖅᒥᑦ	$\mathit{Median}\left(\frac{\lvert C_{t+1}-C_t\rvert}{C_t}\right) \le 0.20$	ᕿᑎᖕᒐᓃᑦᑐᖅ ᕿᑎᖕᒥᑕ ᐊᑯᓐᓂᖅᒥᓐᓃᓪᓚᕆᒃᓗᑎᒃ–ᐊᕐᕌᒍᑕᒪᑦ ᐊᔾᔨᒌᒍᓐᓃᕈᓐᓇᓂᖅᒥᓂᒃ ᐅᓄᓐᓂᖅᒥᑦ ᐱᔭᐅᔪᓐᓇᑦᑎᑕᐅᔪᑦ ᐊᑐᖅᑎᓪᓗᒍ ᐃᓘᓐᓇᓕᒫᖅ ᓴᖅᑭᑎᑕᐅᓂᖅᑕᑕ. ᐱᔪᒥᓇᕐᓂᖅ 20%–ᖅ ᑐᙵᕕᖃᑦᑎᑕᐅᖅᒪᑦ ᒫᓐᓇᕋᑖᖑᓚᐅᖅᑐᖅ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐊᐅᓚᕐᑎᓂᖅ ᐸᕐᓇᒍᑎᓕᐊᕆᓯᒪᔪᒥᒃ ᖃᐅᔨᓴᖅᓯᓂᕐᒧᑦ (MSE) ᐱᓕᕆᔾᔪᓯᖅ.
		ᐃᖃᓗᒃᑕᐅᔪᑦ ᐊᑐᕐᑎᓪᓗᒍ ᐊᖏᓂᖅᐹᒥᒃ ᑕᓯᐃᕙᔅᓴᐅᓂᖅᒪᓂᒃ ᐱᕙᖃᓐᓂᖅ	$\frac{\sum_{1:t_{max}} C_y}{t_{max}}$	ᕿᑎᖅᒐᓂᑦᑐᑦ ᐱᔭᐅᖃᑦᑕᓪᓗᐋᑕᓯᑦ ᐊᑐᖅᑎᓪᓗᒍ ᐱᕙᖃᕐᓂᖅ ᑐᕌᖅᒐᔪᖅ ᐊᖏᓛᒥᒃ ᑕᓯᐃᕙᒃᓴᐅᓂᖅ ᐱᕙᖃᑲᐃᓐᓇᕐᑎᓪᓗᒍ. ᐊᖏᓛᒥᒃ ᑕᓯᐃᕙᔅᓴᐅᓂᖅ ᐱᕙᖃᑲᐃᓐᓇᖅᑎᓪᓗᒍ (t_{max}) ᐊᑯᓂᐅᓂᖅᐹᖑᖅᒪᑦ ᐱᕙᖃᓂᕐᒥ ᖃᐅᔨᔭᐅᔪᑉ ᐃᓘᓐᓇᖅᒐᓂ HCR ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖅᑭᑦ (ᑐᑭᓕᐅᑎᓗᒍ ᖁᓪᓕᖅ ᓯᓇᕐᔪᐊ, ᕿᑎᖅᒐᓂᑦᑐᖅ, ᐊᒻᒪᓗ ᐊᒃᐸᓯᖅᓂᖅᓴᖅ ᓯᓇᕐᔪᐊ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) HCR ᐅᖃᐅᔭᖅ) ᑕᓯᖅᑕᐅᕐᑯᓪᓗᒍ ᐱᕙᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐃᒪᓐᓇᐅᕐᑯᓪᓗᒍ $B_{trigger}$.

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2025 ᐊᒻᒪ 2026 ᐊᑯᑭᑦᑐᓂ ᖃᓕᕋᓖᑦ (GHL) ᑲᑎᕈᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAC)
ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔩᖏᓐᓂᑦ (EAGSAC)
ᐃᓚᒋᔭᐅᔪᓂ ᐃᓱᒪᒋᔭᐅᔪᑦ
ᑲᑎᒪᓂᕐᓂᑦ ᐅᖃᖃᑎᒌᒍᑕᐅᔪᑦ
ᓇᐃᓈᖅᓯᒪᔪᑦ
ᐋᒡᒋᓯ 14, 2024

ᐃᒃᓯᕙᐅᑕᖅ – ᔮᐊᕝ ᐋᑕᒻ, ᐊᕕᒃᑐᖅᓯᒪᔪᓂ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᔨᒻᒪᑎᒃ, ᐅᑭᐅᖅᑕᖅᑐᒥ ᐊᕕᒃᑐᖅᓯᒪᔪᓂ, ᐃᒪᕐᒥᐅᑕᓕᕆᔾᔪᑎᒃᑯᑦ ᑲᓇᑕᒥ (DFO)

ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔩᖏᑦ ᐃᓚᒋᔭᐅᔪᓂᑦ ᐃᓚᒋᐅᖅᑕᐅᔪᑦ

ᑮᓇᐅᔭᓕᐅᕋᓱᓐᓂᕐᒧᑦ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ:
- ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᖃᑐᔾᔨᖃᑎᒌᑦ (NFA) – ᑎᓇᒃ ᕙᑦᑐᖅ, ᑭᒡᒐᖅᑐᐃᔪᖅ:
  - ᐅᑭᐅᖅᑕᖅᑐᒥ ᐃᖅᑲᓗᒐᓯᐊᖅᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᑦ (AFA)
  - ᖁᑭᖅᑖᓗᖕᒥ ᐃᖅᑲᓗᓕᕆᔩᑦ (BFC)
  - ᑎᓗᔾᔭᖅᕙᒻᒥ ᐃᖅᑲᓗᓕᕆᔨᒃᑯᑦ ᑭᒡᓕᓕᓯᒪᔪᑦ (CSFL)
  - ᖁᑭᖅᑖᓗᒃ ᑯᐊᐳᕆᓴᒃᑯᑦ (QC)
- ᐅᑭᐅᑕᖅᑐᒥ ᑲᑐᔾᔨᖃᑎᒌᑦ (NC) – ᐋᓕᔅᑎᐅ ᐅᕋᐃᓕ, ᑭᒡᒐᖅᑐᐃᔪᖅ:
  - ᒪᑭᕝᕙᒃ ᑯᐊᐳᓈᔅᓐ
  - ᓛᐸᑐᐊ ᐃᖅᑲᓗᒐᓯᐊᖅᑏᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑭᖑᑉᐸᓕᕆᓂᕐᒧᑦ ᑎᒥᖓᑦ (Labrador Fishermen’s Union Shrimp Company Ltd.)
  - ᑑᕐᖓᐃᑦ ᐃᖅᑲᓗᓕᕆᔨᒃᑯᑦ ᑯᐊᐸᒃᑯᑦ ᑎᒥᖓᑦ (Torngat Fish Producers Co-operative Society Ltd.)
  - ᓄᓇᑦᓯᐊᕗᑦ ᑎᒥᖅᑯᑎᖏᑦ (Nunatsiavut Group of Companies)
- ᐋᑦᓚᓐᑏᒃ ᒍᕋᐅᓐᕙᔅ ᑲᑎᒪᔩᑦ (AGC) – ᓯᑎᕙ ᑎᕙᑦ, ᑭᒡᒐᖅᑐᐃᓪᓗᓂ:
  - ᐄᑯᓴᐅᓐ ᐃᖅᑲᓗᓕᕆᔩᑦ (EcoSound Fisheries)
  - ᐅᐃᐅᓱᒃ ᐃᖅᑲᓗᓕᕆᔩᑦ
  - Hᐊᑉᐳ ᒍᕋᐃᔅ ᑭᖑᑉᐸᓕᕆᓂᕐᒧᑦ ᑎᒥᖓᑦ (Harbour Grace Shrimp Co. Ltd)
  - ᐆᓴᓐ ᓱᐃᔅ ᐃᓐᑐᓈᓴᓄ (Ocean Choice International)
  - ᒨᕐᓯ ᑕᓇᐅᕐᒥᐅᑕᐃᑦ ᓂᖀᑦ ᑎᒥᖓᑦ (Mersey Seafoods Ltd)
  - ᑭᓖᓐᕗᐊᑕ ᑕᓇᐅᕐᒥᐅᑕᐃᑦ ᓂᖀᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd) (CW)
  - ᓄᐊᑎᒃ ᑎᒥᖓᑦ (Nordic Ltd.)

ᓂᐅᕿᑎᒃᓴᓄᑦ ᐃᖅᑲᓗᒐᓱᖕᓂᖅ ᓂᕿᒃᓴᖅᓯᐅᖅᓂᖅ
- ᖁᑭᖅᑖᓗᒃ ᑯᐊᐳᕆᓴᒃᑯᑦ – ᔮᐊᓇ ᐅᐊᑦ
- ᑭᓖᓐᕗᐊᑕ ᑕᓇᐅᕐᒥᐅᑕᐃᑦ ᓂᖀᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd [CW]) – ᑳᑐᓇᓐ ᐳᐃᑦ

ᒐᕙᒪᒃᑯᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ
- ᒐᕙᒪᖕᒋᑦ ᓂᐅᕗᓐᓛᓐ ᐊᒻᒪᓗ ᓛᐸᑐᐊ – ᔭᐃᒃ ᕋᐃᔅ
- ᐃᒪᕐᒥᐅᑕᓕᕆᔾᔪᑎᒃᑯᑦ ᑲᓇᑕᒥ – ᑭᕙᓐ Hᐃᔾᔨ, ᖃᐅᔨᓴᓂᕐᒧᑦ ᖃᐅᔨᓴᖅᑎ
- ᐃᒪᕐᒥᐅᑕᓕᕆᔾᔪᑎᒃᑯᑦ ᑲᓇᑕᒥ – ᑲᐃᑦ ᔫᓐᓇᓐ, ᓯᓚᕐᔪᐊᒥ ᐃᖅᑲᓗᓕᕆᓂᕐᒧᑦ ᐊᑐᐊᒐᐃᑦ

ᐊᕙᑎᓕᕆᓂᕐᒧᑦ ᒐᕙᒪᐅᙱᑦᑐᓂ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ
- ᐃᒫᓗᒃ ᑐᓄᕙᐊᖅ – ᐳᓇᓐ ᑕᕙᓐ

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ᑕᑯᓐᓇᖅᑏᑦ

- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) – ᓵᓐᑐᕋ ᒧᐊ (FM-AR), ᐊᓕᒃᓯᔅ ᔼᑦ (FM-AR), ᑑᒃ ᐃᐅᔨᕋ (RM-NCR)

ᑲᑎᒪᓂᐅᑉ ᐱᔾᔪᑎᖓ:

ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᕚᓄᑦ (EAGSAC) ᐃᓚᒋᔭᐅᔪᓄᑦ ᑐᓴᕐᑲᑎᒋᒍᑎᕐᑲᖅᓯᒪᔪᑦ ᐃᓱᒪᒋᔭᖏᓐᓂᒃ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐊᓕᒃᑲᓐᓂᖅ ᐃᓕᓂ 0 (GHL) ᑲᑎᑲᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAC) ᖃᓄᑎᒋᐅᓂᖏᑦ 2025-ᒧᑦ ᐊᒻᒪᓗ 2026-ᒧᑦ. ᑲᑎᒪᓂᖅᑕᐅᖅ ᐱᕕᒃᓴᐅᓯᒪᔪᖅ ᑲᓇᕐᓇᖓᓂ ᐃᖅᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᕚᓄᑦ (EAGSAC) ᐃᓚᒋᔭᐅᔪᑦ ᐊᐱᖅᓱᕈᓐᓇᖅᑯᑎᒃ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐊᐱᖅᑯᑎᓂᒃ ᐊᒻᒪᓗ ᑐᑭᓯᒋᐊᒃᑲᓐᓂᕐᓂᕐᒥᒃ GHL SA0+1-ᒧᑦ (ᐃᒪᕙᖕᒥ) ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔾᔪᑦ (SC) ᐅᖃᐅᔾᔨᕆᐊᕈᑎᓄᑦ 2025-ᒧᑦ ᐊᒻᒪᓗ 2026-ᒧᑦ.

ᐅᖃᐅᓯᐅᓗᓂ:

ᑭᒡᓕᓯᓂᐊᖅᓯᒪᔪᑦ ᐅᖃᐅᔾᔨᕆᐊᕈᑏᑦ

ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᖅᑏᑦ (ᑭᕙᓐ Hᐃᔾᔨ) ᑭᐅᓚᐅᖅᑐᖅ ᓱᖅᑯᐃᓇᖅᓯᑎᕆᓪᓗ ᐱᖃᖃᑕᐅᔪᑦ ᐊᐱᖅᑯᑎᖏᑦ ᐅᖃᐅᓯᒃᓴᖏᓪᓗ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔾᔪᑦ (SC) ᐅᖃᐅᔾᔨᕆᐊᕈᑎᖏᓐᓄᑦ.

- ᑭᓐᓈᕗᐊᑕ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓂᕿᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd [CW]) ᐊᐱᕆᓚᐅᖅᑐᑦ ᑐᓴᐅᒪᔾᔪᑎᒃᑲᓐᓂᕐᓄᑦ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᓂᕈᐊᖅᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᖅᑭᒃᑯᐊᖅᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖕᒋᓐᓇᖅᑐᒃᑯᑦ (SPiCT ᐋᖅᑭᒃᑯᐊᖅᓯᒪᔪᖅ) ᐊᑐᖅᑕᐅᔪᖅ ᐱᕙᓪᓕᐊᑎᑎᓂᕐᒧᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐅᖃᐅᔾᔨᕆᐊᕈᑏᑦ, ᓛᓐᓇ ᓯᕗᓂᒃᓴᒥᓪᓗ ᑎᑎᖅᑲᖁᑏᑦ ᐊᑐᖅᑕᐅᓂᖏᓐᓄᑦ ᐋᖅᑭᒃᑯᐊᖅᑕᐅᓯᒪᔪᒧᑦ, ᐊᒻᒪᓗ ᓄᑖᑦ ᑭᖑᓖᑦ ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᕿᒥᕐᕈᓂᕐᒧᑦ.
  - ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᖅᑏᑦ ᑭᐅᓯᒪᔪᑦ ᑖᒪᓐᓇ ᓂᕈᐊᖅᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᖅᑭᒃᑯᐊᖅᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖕᒋᓐᓇᖅᑐᒃᑯᑦ (SPiCT ᐋᖅᑭᒃᑯᐊᖅᓯᒪᔪᖅ) ᐊᑐᓕᖅᑕᐅᓯᒪᔪᖅ ᐊᒻᒪ ᓈᒻᒪᒋᔭᐅᓪᓗᓂ ᑖᒃᑯᓇᖕᖓᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) ᔪᓂᐅᓵᖅᑐᒥ. ᐊᕐᕋᒍᑕᒫᑦ ᓄᑖᑦ ᖃᐅᔨᓴᕈᑏᑦ ᐱᓕᕆᐊᖑᖃᑦᑕᕐᓂᐊᖅᑐᑦ ᑕᒪᐃᓐᓂᒃ ᑲᓇᑕᒥ ᐊᒻᒪᓗ ᐊᑯᑭᑦᑐᓂ (ᐃᑦᓵᒃᑰᕐᓗᑎᒃ) ᐊᒻᒪ ᓄᑖᑦ ᑎᑎᖅᑲᖁᑏᑦ ᐃᓕᐅᖅᑲᖅᑕᐅᓂᐊᖅᑐᑦ ᓂᕈᐊᖅᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᖅᑭᒃᑯᐊᖅᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖕᒋᓐᓇᖅᑐᒃᑯᑦ (SPiCT) ᐋᕿᒃᑯᐊᖅᑕᐅᓯᒪᔪᒧᑦ. ᐱᑕᖃᖅᑐᖅ ᑎᓴᒪᑦ ᐊᕐᕌᒍᓄᑦ ᐊᖏᕈᒻᒥᒃ ᐊᑯᑭᑦᑐᒥ ᑲᓇᑕᒧᑦ ᐊᑐᐃᓐᓇᕐᑯᓪᓗᑎᒃ *ᕿᒥᕐᖁᓂᕐᒧ ᐅᒥᐊᕐᔪᐊᖅ* Tarajoq ᐊᕐᕋᒍᑕᒫᑦ ᖃᐅᔨᓴᕈᓪᓗᒍ SA0-ᒥ.
  - ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖏᑦ ᐋᖅᑭᒃᑯᐊᖅᑕᐅᓯᒪᔪᓂᒃ-ᑐᙱᕙᓖᑦ ᓱᒃᑲᓕᕙᓪᓕᐊᓂᖏᑦ ᑖᒃᑯᓇᓂ *ᕿᒥᕐᖁᓂᕐᒧ ᐅᒥᐊᖅᔪᐊᖅ* Paamiut ᐊᒻᒪᓗ *ᕿᒥᕐᖁᓂᕐᒧ ᐅᒥᐊᖅᔪᐊᖅ* Tarajoq ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᖃᖓᒃᑰᕐᑲᑦᑕᕐᓂᖏᑦ ᑲᔪᓯᕙᓪᓕᐊᖕᒋᓐᓇᖅᑐᖅ, ᐊᒻᒪᓗ ᐱᔭᕇᖅᑕᐅᒍᑎᒃ ᐃᑲᔪᕐᓂᐊᖅᑐᑦ ᐃᑕᖅᑲᕐᓂᖏᓐᓄᑦ ᕿᒥᕐᕈᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᓇᓱᒃᑐᓂᒃ. ᐱᓇᓱᐊᖅᓂᐊᕐᑐᑦ ᓄᑖᙳᖅᑎᕆᓗᑎᒃ ᐊᑕᐅᓯᖕᒥᑦ ᑭᖑᓕᕐᓂᑦ ᒪᓕᒡᓗᑎᒃ.
  - ᐅᕿᐊᓵᖓᓂ 2024 ᖃᐅᔨᓴᕐᓂᕐᒥ ᖃᐅᔨᓴᕈᑏᑦ ᕿᒥᕐᕈᔭᐅᓚᐅᖅᑐᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) ᒪᐃ 2025-ᖑᑎᓪᓗᒍ, ᐊᑐᖅᑕᐅᓪᓗᑎᒃ ᓂᕈᐊᖅᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᖅᑭᒃᑯᐊᖅᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖕᒋᓐᓇᖅᑐᒃᑯᑦ (SPiCT) ᐋᖅᑭᒃᑯᐊᖅᑕᐅᓯᒪᔪᓂ ᑲᑎᑲᒋᑦ ᐊᑯᑭᑦᑐᓂ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ, ᐋᖅᑭᒃᑯᐊᖅᑕᐅᓯᒪᔪᖅ ᓴᖅᑭᑕᐅᓂᐊᖅᑐᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) ᑲᑎᒪᓕᕈᑎᒃ ᔪᓂ 2025-ᒥ. ᓄᑖᖑᓖᑦ ᐅᓂᒃᑳᑦ ᓴᖅᑭᑕᐅᔪᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓂᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) 2025-ᒥ, ᑭᖑᓪᓖᑦ ᐊᑕᖑᐃᓯᒪᔪᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ

ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓂᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) ᐱᑕᕐᑲᕐᓂᖅᑭᓐᓄᑦ ᕐᑭᒥᕐᕈᓂᕐᒃ ᐅᑦᑐᒃᓴᕐᑲᕐᒪᑦ ᐱᓕᕆᐊᖑᓛᕐᒍᓂ 2026-ᒥ.

- ᓄᓇᕗᒻᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᕐᑲᑐᔾᔨᕐᑲᑎᒌᑦ (NFA) ᑕᐃᓯᓯᒪᔪᑦ ᔪᓚᐃ 8 ᐅᓂᒃᑳᕐᑕᐅᓚᐅᕐᑐᓂᒃ ᓴᕐᑭᑖᐅᕐᕕᓂᒃᓯᒪᙱᑦᑑᑦᑐᑎᒃ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓂᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) ᐅᕐᑲᐅᔾᔨᕆᐊᕈᑏᑦ ᑐᓂᔭᐅᓯᒪᔪᑦ ᑲᓇᕐᓇᖕᑲᓂ ᐃᕐᑲᕐᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᕐᑲᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕐᕿᓛᓄᑦ (EAGSAC) ᐃᓚᒋᔭᐅᔪᓄᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᑎᖅᑭᓐᓂᒃ. ᓄᓇᕗᒻᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᕐᑲᑐᔾᔨᕐᑲᑎᒌᑦ (NFA) ᐊᐱᕆᓯᒪᔪᑦ ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᔭᐅᓯᒪᔪᑦ ᐃᕐᑲᓗᓗᐋᔾᔭᕐᓂᕐᒧᑦ ᐅᔾᔨᕐᓱᕐᓂᒃᑯᑦ ᐊᑐᕐᑕᐅᓇᓯᐊᕐᑐᑦ (PA) ᐋᕐᑭᒃᓯᒪᓂᖅᑭᑦ (ᓴᕐᑭᑎᑕᒐᕐᒃ 7) ᐃᓱᒪᒋᕐᑲᓯᐅᔾᔭᐅᕐᔪᐊᕐᓯᒪᔪᑦ. ᓄᓇᕗᒻᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᕐᑲᑐᔾᔨᕐᑲᑎᒌᑦ (NFA) ᕐᑲᐅᔨᕆᐊᕐᓯᒪᔪᑦ ᕐᑲᓄᕐᒃ ᑖᒃᑯᐊ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᕐᓂᒃᑯᑦ ᐊᑐᕐᑕᐅᓇᓯᐊᕐᑐᑦ (PA) ᐋᕐᑭᒃᓯᒪᓂᖅᑭᑦ, ᒫᓇ ᕐᑭᒥᕐᕆᕈᔭᐅᕙᓪᓕᐊᔪᑦ, ᐊᔾᔨᙱᖕᓇᓴᕐᒪᖔᖅᑭᑦ ᑲᓇᑕᒥ ᐅᔾᔨᕐᓱᕐᓂᒃᑯᑦ ᐊᑐᕐᑕᐅᓇᓯᐊᕐᑐᑦ (PA) ᐋᕐᑭᒃᓯᒪᓂᖅᑭᓐᓄᑦ, ᐊᒻᒪ ᕐᑲᓄᑐᐃᓐᓇᕐᒃ ᓄᑖᖑᕆᐊᕈᑎᑕᕐᑲᕐᒪᖔᑦ ᕐᑭᒥᕐᕈᓂᕐᒥ ᐊᑐᐃᓐᓇᐅᔪᕐᑲᕐᒪᖔᑦ.
  - ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᑎᖅᑭᑦ ᐅᕐᑲᕐᓯᒪᔪᑦ ᐱᓕᕆᓂᕐᒧᑦ ᓈᓴᐅᓯᕆᔾᔪᑏᑦ ᐃᓱᒪᒋᕐᑲᓯᐅᔾᔭᐅᓯᒪᙱᑦᑐᑦ, ᑖᒃᑯᐊ ᐱᖕᒐᓯᑦ ᓇᓕᒧᒌᑦ ᒪᓕᒃᑕᐅᔪᒃᓴᑦ ᐊᑐᕐᑕᐅᑦᑐᑎᒃ ᐊᑦᑕᕐᓇᙱᒃᑐᒃᑯᕕᐅᑦᑐᑎᒃ.
  - ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᑎᖅᑭᑦ ᐅᔾᔨᕈᓱᙱᑦᑐᑦ ᓄᑖᖑᕆᐊᕈᑎᑕᕐᑲᕐᒪᖔᑦ ᒫᓇᐅᔪᕐᒃ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᕐᓂᒃᑯᑦ ᐊᑐᕐᑕᐅᓇᓯᐊᕐᑐᑦ (PA) ᐋᕐᑭᒃᓯᒪᓂᖅᑭᓐᓂᒃ ᕐᑭᒥᕐᕈᓂᕐᒥ. ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᕐᓂᒃᑯᑦ ᐊᑐᕐᑕᐅᓇᓯᐊᕐᑐᑦ (PA) ᐋᕐᑭᒃᓯᒪᓂᐅᔪᕐᒃ ᒪᓕᖕᓂᕐᓴᐅᓕᑐᐃᓐᓇᑎᐊᓕᒃ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᐅᔾᔨᕐᓱᕐᓂᒃᑯᑦ ᐊᑐᕐᑕᐅᓇᓯᐊᕐᑐᑦ (PA) ᐋᕐᑭᒃᓯᒪᓂᖅᑭᓐᓂᒃ.

- NC–ᑯᑦ ᐅᕐᑲᕐᓯᒪᔪᑦ ᑖᒪᓐᓇ ᓂᕈᐊᕐᑕᐅᑐᐃᓐᓇᕐᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᕐᑭᙳᐊᕐᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖅᑭᓐᓇᕐᑐᒃᑯᑦ (SPiCT) ᐋᕐᑭᙳᐊᕐᓯᒪᔪᕐᒃ ᓄᑖᖑᔪᕐᒃ, ᓇᓗᓇᐃᕐᑕᐅᓯᒪᙱᑦᑐᕐᒃ ᓯᓚᔫᓂᖑᓂᒃ, ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᑎᑎᕐᑲᕐᑯᑎᕐᑲᑦᑎᐊᖅᑭᓐᓂᖑᓄᑦ (ᐱᕐᑲᑎᐅᑎᓪᓗᒋᑦ ᕐᑲᐅᔨᓴᕈᑏᑦ ᐃᓚᑰᓂᖅᑭᓐᓄᑦ ᕐᑲᖑᒃᑰᕐᑲᑦᑕᕐᓂᖅᑭᑦ), ᐊᑯᑦᑐᓗᐊᕐᓗᑎᒃ ᕐᑯᓯᙱᑦᑎᐊᕐᓂᖅᑭᑦ, ᐊᒻᒪ ᐱᕐᑲᓯᐅᔾᔨᓯᒪᓪᓚᓗᐊᕐᑎᓪᓗᒍ ᐊᖑᓂᒃᑕᐅᕙᒃᑐᑦ ᕐᑲᐅᔨᓴᕈᑎᖅᑭᓐᓂᒃ, ᐊᖑᓂᒃᑕᐅᔪᑦ ᐊᑐᓂ ᐊᖑᓇᓱᒃᑎᓪᓗᒋᑦ (CPUE) ᓴᓇᕝᕕᐅᔪᒥ ᐱᕐᑲᓯᐅᔾᔭᐅᓯᒪᙱᑦᑐᑦ, ᑕᒪᓐᓇᓗ ᐊᖅᑭᔫᑐᐃᓐᓇᑎᐊᓕᒃ. ᐊᑕᖅᑭᕐᒍᑎᑦ ᑕᒪᒃᑯᐊ ᑎᑭᐅᑎᓯᒪᔪᑦ ᓇᓗᓇᕐᑑᑦᑐᓂ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐅᕐᑲᐅᔾᔨᕆᐊᕈᑎᓄᑦ. NC-ᑯᑦ ᐊᐱᕆᓯᒪᔪᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᑎᖅᑭᓐᓂᒃ ᑲᒪᕐᑯᑦᑐᑎᑦ ᑕᒪᒃᑯᓂᖓ ᐃᓱᒫᓘᑕᐅᔪᓂᒃ.
  - ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᑎᖅᑭᓐᓂᒃ ᑭᐅᓚᕗᕐᑐᑦ ᑕᒪᓐᓇ ᕐᑯᓯᙱᑦᑎᐊᕐᓂᖓ ᐊᑯᑦᑐᓂᖅᑭᑦ ᐊᖅᑭᔫᖅᒪᑕ ᑭᓯᐊᓂᓕ ᓈᒻᒪᒋᔭᐅᓐᓄᑎᒃ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC). ᐱᑕᕐᑲᙱᑦᑐᑦ ᐊᐃᕙᔾᔪᑎᓂᒃ ᑕᒪᑐᒧᖅᑕ ᕐᑯᓯᙱᑦᑎᐊᕐᓂᕐᒧᑦ ᐋᕐᑭᙳᐊᕐᑕᐅᓯᒪᙱᑕᓂᒃ, ᐋᕐᑭᑦᑎᐊᕐᓯᒪᓂᖅᑕᓄᑦ ᐊᒻᒪ ᓈᒻᒪᒋᔭᐅᑦᑐᓂ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC). ᕐᑯᓯᙱᑦᑎᐊᕐᓂᕐᒃ ᕐᑲᐅᔨᓴᕈᑎᓂᒃ ᐱᕙᓪᓕᐊᓂᐊᕐᑐᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ ᑕᑭᓂᖅᑭᑦ ᐊᒻᒪ ᐅᑭᐅᖅᑭᓐᓄᑦ ᕐᑲᐅᔨᓴᕈᑏᑦ ᑲᑎᑕᐅᓕᕐᐸᑕ. ᕐᑲᐅᔨᓴᕈᑎᒃᑲᓐᓃᑦ ᐃᓚᔭᐅᕙᓪᓕᐊᓂᐊᕐᑐᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ, ᐊᒻᒪ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐃᕐᑲᓇᐃᔭᕐᕕᐃᑦ ᕐᑭᓅᓇᐅᔭᕐᑐᑦ ᐱᐅᓯᕆᐊᕈᑎᒃᓴᓂᒃ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᔪᓄᑦ ᐊᒻᒪ ᐋᕐᑭᙳᐊᕐᑕᐅᓯᒪᔪᓄᑦ.

- NC-ᑯᑦ ᕐᑲᐅᔨᕆᐊᓚᐅᕐᑐᑦ ᐊᐱᕆᓪᓗᑎᒃ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᒃᐸᓯᒃᑑᓂᖑᓂᒃ ᐅᓗᕆᐊᓇᕈᓐᓇᑐᐃᓐᓇᕆᐊᓖᑦ (ᒫᓇ ᐊᖑᓂᒃᐸᒃᑐᑦ ᕐᑲᓄᑎᒋᐅᓂᖅᑭᑦ ᒪᓕᒃᓗᒋᑦ), ᐊᖑᓇᓱᖕᓃᑦ ᕐᑲᓄᑎᒋᐅᓂᖅᑭᑦ ᐅᓄᕈᓐᓃᓇᓂᐊᕐᑎᑕᐅᔪᓐᓇᕐᑐᑦ ᐱᔾᔪᑕᐅᓗᑎᒃ ᐊᖑᓇᓱᖕᓃᑦ ᐃᓚᒌᓗᐊᕐᑎᑦᓗᒋᑦ, ᐱᔾᔪᑕᐅᑦᓗᑎᒃ ᐋᕐᑭᙳᐊᕐᓯᒪᔪᑦ ᑐᖑᕕᑦᑎᓯᒪᖕᒪᑕ ᐊᑐᒃᑲᓂᕈᓇᕐᑐᑦ ᓄᓇᒥᖕᖑᕐᑐᑦ ᐊᒻᒪᓗ ᐅᖑᑕᐅᔾᔨᓯᒪᓪᓗᑎᒃ ᑐᕐᑯᑕᐅᕐᑲᑦᑕᕐᑐᓃᒃ? NC-ᑯᑦ ᓇᓗᓇᐃᕐᓯᓚᐅᕐᑐᑦ ᕐᑲᐅᔨᓴᕈᑎᓂ ᕐᑲᐅᔨᔭᐅᔪᑦ ᐊᑑᓯᕐᑐᓂ ᒪᕐᕉᖕᓂ ᐊᕐᕌᒍᖕᓂ ᐱᐅᓗᐊᙱᒪᑕ, ᐊᒻᒪᓗ ᓇᓗᓇᐃᕐᓯᓯᒪᙱᓗᑎᒃ “ᐅᔾᔨᓱᓇᐊᓕᖕᓂᒃ”.
  - ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᕐᑎᖅᑭᑦ ᑭᐅᓚᐅᕐᑐᑦ ᐱᖕᒐᓯᓂᒃ ᒪᓕᒃᑕᐅᔪᒃᓴᓂᒃ

ᐃᓱᒪᒋᖃᓯᐅᔾᔨᓯᒪᔪᑦ ᓇᓗᓇᐃᖅᑕᐅᖁᓪᓗᒋᑦ ᐊᑦᑕᕐᓇᙱᑦᑐᒃᑯᓈᑦ ᐱᑕᖃᕕᐅᔪᓄᑦ, ᐃᒪᐃᑐᐃᓐᓇᕆᐊᖃᖅᑐᓄᑦ $F>F_{lim}$ (ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᓖᑦ ᐃᖃᓗᒐᓱᓗᐊᕐᓂᖅᑕᖃᕐᓗᓂ, ᐅᓄᕈᓐᓃᕐᓗᑎᒃ ᐱᑕᖃᕕᐅᔪᑦ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖒᖅᑐᑦ ᓂᕆᐅᒋᔭᐅᓇᔭᖅᑐᑦ), ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖏᑦ $B<B_{lim}$ (ᖃᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖏᓐᓄᑦ ᐱᕐᔪᐊᖑᓪᓗᓂ ᐃᓂᐅᔪᑦ ᑕᐃᑲᓂ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ [DFO] ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ [PA] ᐋᖅᑭᒃᓯᒪᓂᖏᑦ), ᐊᒻᒪ ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ $B>B_{msy}$ (ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᓖᑦ ᐅᖓᑖᓃᓪᓗᑎᒃ B_{msy}, ᐅᓄᓛᒥᒃ ᐃᕐᓂᐅᕐᓂᖏᑦ ᑎᑭᑕᐅᒍᑎᒃ, ᐊᑐᖅᑕᐅᓇᔭᖅᑐᖅ ᑐᑭᓯᑎᑕᐅᓗᑎᒃ ᖁᑦᑎᖕᓂᖅᓴᑦ ᑭᒡᓕᐅᔪᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᑕᐃᑲᓂ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ [DFO] ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ [PA] ᐋᖅᑭᒃᓯᒪᓂᖏᓐᓂᑦ). ᐊᖑᓂᒃᑕᐅᔪᓗᒃᑖᐸᓗᐃᑦ ᖃᓄᑎᒋᐅᓂᖏᓐᓄᑦ ᐋᖅᑭᙳᐊᖅᑕᐅᓯᒪᔪᑦ ᐅᖓᑖᓅᒃᑯᑎᒃ 30% ᖃᓄᐃᓪᓕᔪᖃᖅᑳᖅᑎᓐᓇᒍ ᑭᒡᓕᐅᔪᓄᑦ ᑕᒪᑐᒧᖓ $F>F_{lim}$. ᑎᑎᕋᖅᓯᒪᔪᖅ ᑕᐃᑲᓂ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ (SC) ᐅᖃᐅᔾᔨᒋᐊᕈᑎᒥ ᐊᑐᐃᓐᓇᐅᑎᑦᑎᔪᖅ ᐅᓗᕆᐊᓇᕈᓐᓇᖅᑐᓂᒃ-ᑐᙵᕕᖃᖅᖢᑎᒃ ᐅᖃᐅᔾᔨᒋᐊᕈᑎᒥᒃ. ᐊᑕᐅᓯᖅ ᑐᑭᓕᐊᖅᑐᖅ ($F>F_{lim}$ <30%) "ᐃᓄᑐᐊᖑᑎᑕᐅᔪᖅ" ᐊᒻᒪ ᐊᑐᖅᑕᐅᓪᓗᓂ "ᓯᐊᕐᓇᒥᒃ ᑕᖅᓴᓕᒃ ᑭᑉᐹᕆᒃᑐᖅ", ᑭᓯᐊᓂᓕ ᑐᒪᓴᐅᒪᔾᔪᑏᑦ ᐊᑐᐃᓐᓇᕈᖅᑕᐅᔪᑦ ᐊᔾᔨᒌᙱᑦᑑᔭᕐᓂᖏᓐᓄᑦ ᐊᖏᓂᒃᑕᐅᕙᒃᑐᑦ ᖃᓄᑎᒋᐅᓂᖏᓐᓄᑦ ᐋᖅᑭᙳᐊᖅᑕᐅᓯᒪᔪᓄᑦ.

  - ᖃᓄᑎᒋᐅᓂᖓ ᐅᓗᕆᐊᓇᕈᓐᓇᕐᓂᖓ ᑭᒡᓕᓯᓂᐊᕐᓂᒃᑯᑦ ᐃᓱᒪᓕᐅᕈᑕᐅᙱᑦᑐᖅ, ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐊᐅᓚᑦᑎᔩᑦ ᐃᓱᒪᓕᐅᓲᑦ ᑕᒪᑐᒥᖓ. ᐅᓄᕐᓂᖅᓴᐅᒍᑎᒃ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ, ᐅᓄᕐᓂᖅᓴᐅᔭᓐᓇᖅᑐᑦ ᐱᑕᖃᕐᓂᐅᔪᑦ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖒᖅᑐᑦ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕐᓂᐊᖅᑐᑦ. ᐊᖑᓂᒃᑕᐅᔪᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ ᐋᖅᑭᒃᑕᐅᓯᒪᒍᑎᒃ 75% F_{msy} ᖃᓄᐃᓪᓕᔪᖃᖅᑳᖅᑎᓐᓇᒍ ᑭᒡᓕᐅᔪᓄᑦ, ᑕᒪᓐᓇᓕ ᑕᐃᒪ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᓪᓗᐊᖅᑐᑦ ᓂᕆᐅᒋᔭᐅᓗᑎᒃ ᓯᕗᓂᒃᓴᒥ. ᐃᓱᒪᒋᔭᐅᓪᓗᐊᑕᓇᐊᓕᒃ ᖃᓄᑎᒋᑦ ᓱᕐᕌᕐᑯᔭᐅᙱᓐᒪᖔᑕ ᑖᒃᑯᐊ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ. ᐅᓄᕈᓐᓃᖅᑎᑕᐅᕌᕐᔪᙱᓐᓂᖏᑦ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ ᒫᓐᓇ ᐊᑐᙱᑎᑦᑎᑐᐃᓐᓇᕆᐊᓕᒃ ᐅᓄᕈᓐᓃᕐᔪᐊᓗᐊᙱᓐᓂᕐᒥᒃ ᐊᒥᓲᙱᑦᑐᓂ ᐊᕐᕌᒍᓂ.

- ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᖃᑐᔾᔨᖃᑎᒌᑦ (NFA) ᑕᐃᓯᓯᒪᔪᑦ ᓴᖅᑭᑎᑕᒐᖅ 2-ᒥᒃ ᔪᓚᐃ 8-ᓂᓴᑦ ᐅᓂᒃᑳᖅᑕᐅᔪᑦ ᐊᒻᒪ ᐊᐱᕆᓯᒪᓪᓗᑎᒃ ᐅᓗᕆᐊᓇᕈᓐᓇᖅᑐᓂᒃ ᐱᔾᔪᑎᖃᖅᖢᑎᒃ ᐅᖃᐅᔾᔨᒋᐊᕈᑏᑦ ᓯᕗᓂᒃᓴᒧᑦ.
  - ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑭᒡᓕᓯᓂᐊᖅᑎᖏᑦ ᐅᓂᒃᑳᖅᓯᒪᒧᑦ ᑕᒪᓇ ᑐᒃᓯᕋᖅᑕᐅᑎᓪᓗᒍ ᐅᖃᐅᔾᔨᒋᐊᕈᑎ ᑎᑎᕋᖅᑕᐅᓕᖅᑎᓪᓗᒍ ᐊᒻᒪ ᑐᓂᔭᐅᓪᓗᑎᒃ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓄᑦ (SC) ᓈᒻᒪᑐᖅᑕᖅᑲᖅᓯᒪᙱᒻᒪ ᑕᒪᑐᒧᖓ ᐱᑕᖃᕕᐅᔪᓄᑦ (ᓲᕐᓗ ᓂᕈᐊᖅᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᖅᑭᙳᐊᖅᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖏᓐᓇᖅᑐᒃᑯᑦ [SPiCT]) ᐊᒻᒪ ᑲᖏᕿᔭᐅᓪᓗᓂ ᑕᒪᓐᓇ ᑭᒡᓕᓯᓂᐊᕐᓂᒃᑯᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᑎ ᐱᔾᔪᑎᖃᐃᓐᓇᕐᓂᐊᕐᓂᖓᓂᒃ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᖒᖅᑐᑦ ᐊᒻᒪ ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᓇᓗᓇᐃᕈᑏᑦ. ᓈᒻᒪᒋᔭᐅᑎᓪᓗᒍ ᓂᕈᐊᖅᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᐊᒥᐊᒃᑯᑦ ᐱᕙᓪᓕᐊᓂᕐᒧᑦ ᐋᖅᑭᙳᐊᖅᓯᒪᔪᑦ ᑲᔪᓯᕙᓪᓕᐊᖏᓐᓇᖅᑐᒃᑯᑦ (SPiCT) ᐋᖅᑭᙳᐊᖅᓯᒪᔪᖅ, ᓯᕗᓂᒃᓴᒥ ᐱᑕᖃᕕᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᓐᓄᑦ ᑐᕌᖅᑕᐅᔪᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᔭᓐᓇᖅᑐᑦ, ᐊᒻᒪ ᐅᓗᕆᐊᓇᕈᓐᓇᖅᑐᓂᒃ-ᐱᔾᔪᑎᓖᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᑏᑦ ᐊᑐᐃᓐᓇᕈᖅᑕᐅᔭᓐᓇᖅᑐᑦ.

ᐃᖃᓗᑦᑕᐅᔭᓐᓇᖅᑐᑦ ᑕᑯᓐᓇᖅᑕᐅᔪᑦ ᑐᓂᔭᐅᔪᑦ:

- ᐊᖏᖃᑎᒌᖃᑎᖃᖏᓐᓂᐅᔪᑦ ᐃᖃᓗᑦᑕᐅᔭᓐᓇᖅᑐᓂᑦ ᖁᕝᕙᓯᓐᓂᖓᓗ ᐅᖃᐅᔾᔨᒋᐊᕈᑎ ᐱᑕᖃᖅᑎᑕᐅᔪᖅ ᐅᑯᓄᖓ ᐅᐊᖕᓇᖓᑕ ᐱᖓᓐᓇᓂ ᐊᑦᓛᓐᑎᒃ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᑲᑎᒪᔨᖏᑦ ᐅᕙᓂ 2024 (ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᖃᑐᔾᔨᖃᑎᒌᑦ [NFA], ᐅᑭᐅᑕᖅᑐᒥ ᑲᑐᔾᔨᖃᑎᒌᑦ [NC]):
  - ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᖃᑐᔾᔨᖃᑎᒌᑦ (NFA) ᐊᒻᒪ NC-ᑯᑦ ᑐᒃᓯᕋᑐᐃᓐᓇᕆᐊᓖᑦ ᑲᔪᓯᑎᑕᐅᓗᑎᒃ 2024-ᒥ ᑲᑎᖦᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔭᓐᓇᖅᑐᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ 2025-ᒥ ᐊᒻᒪ 2026-ᒥ ᐅᓄᙱᑖᖑᔭᓐᓇᖅᑐᒃᑯᑦ. NC-ᑯᑦ ᓇᓗᓇᐃᖅᓯᓯᒪᔪᑦ ᓴᖅᑭᑦᑐᖃᑐᐃᓐᓇᕆᐊᓖᑦ ᐅᓄᖅᓯᒋᐊᕆᓂᕐᒥᒃ ᑲᑎᖦᖢᒋᑦ ᐊᑯᓂᒃᑕᐅᔭᓐᓇᖅᑐᓄᑦ ᑭᖑᓂᖓᑦᑎᓐᓂ ᐅᓄᖒᑐᓕᐅᖅᓯᒪᔪᓄᑦ ᖃᓄᑎᒋᐅᓂᖏᓐᓄᑦ.

- ᖃᒃᑲᓐᓂᐊᒍᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐱᓕᕆᓃᑦ ᐅᖃᐅᓯᖃᖅᑎᓪᓗᒋᑦ ᐃᓚᓕᐅᔾᔨᓂᕐᒥᒃ ᖃᐅᔨᓴᕈᑎᒃᑲᓐᓂᕐᓂᒃ ᐊᑐᕐᓗᑎᒡᓗ ᐋᖅᑭᒃᑕᐅᓯᒪᔪᓂᒃ-ᐱᔾᔪᑎᓖᑦ ᐱᕙᓪᓕᐊᓂᐅᔪᓄᑦ ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᕿᒥᕐᕈᓂᕐᒥ ᑐᒃᓯᕋᖅᑕᐅᓯᒪᔪᖅ, ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᐊᑕᖏᖅᓯᓯᒪᔪᑦ ᐱᑕᖃᕕᐅᔪᓂᒃ ᕿᒥᕐᕈᓂᕐᒧᑦ ᑕᐃᑲᓂ ᐅᐊᖕᓇᒥᒃ

ᐃᓚᒍᑕᖓ 4

ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ (SC) ᑲᑎᒪᓂᒃᓴᖓᓂ 2025-ᒥ (ᐋᑦᓚᓐᑏᒃ ᒍᕋᐅᓐᕕᔅ ᑲᑎᒪᔩᑦ, ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᖃᑐᔾᔨᖃᑎᒌᑦ, ᑭᓛᓐᕗᐊᑕ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓂᖀᑦ ᑎᒥᖓᑦ [Clearwater Seafoods Ltd], ᕿᑭᖅᑖᓗᒃ ᑯᐊᐳᕆᓴᒃᑯᑦ) ᐅᓪᓗᒃᓴᖏᓐᓂᑐᐃᓐᓇᐅᙱᑦᑐᖅ ᓖᓐᓇᓚᐅᑲᒃ ᑐᓴᒐᒃᓴᓄᑦ.

- ᐋᑦᓚᓐᑏᒃ ᒍᕋᐅᓐᕕᔅ ᑲᑎᒪᔩᑦ (AGC), ᐅᑭᐅᑕᖅᑐᒥ ᑲᑐᔾᔨᖃᑎᒌᑦ (NC), ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᖃᑐᔾᔨᖃᑎᒌᑦ (NFA), ᐊᒻᒪᓗ ᒐᕙᒪᖓᑦ ᓂᐅᕗᓐᓛᓐ ᐊᒻᒪᓗ ᓛᐸᑐᐊ ᐊᑐᓂᓯᓂᐊᖅᑐᑦ ᑎᑎᕋᖅᓯᒪᔪᓂᒃ ᐃᒪᕐᒥᐅᑕᓕᕆᔩᒃᑯᑦ ᑲᓇᑕᒦᒃᑯᓐᓄ (DFO) ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ 2025-ᒧᑦ ᐊᒻᒪᓗ 2026-ᒧᑦ.

- ᑭᓛᓐᕗᐊᑕ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓂᖀᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd) ᐊᖏᑐᐃᓐᓇᕆᐊᔨᓛᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖃᓄᑎᒋᐅᓂᖏᓐᓄᑦ ᑐᓂᔭᐅᔪᑦ ᑖᒃᑯᓇᙵᑦ ᐋᑦᓚᓐᑏᒃ ᒍᕋᐅᓐᕕᔅ ᑲᑎᒪᔨᓂᑦ (AGC). ᖄᒃᑲᓐᓂᐊᒍᑦ, ᑐᙵᕕᒋᓪᓗᒋᑦ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐱᓕᕆᓂᒃᑲᓐᓃᑦ (ᑲᑎᖅᓯᐃᓃᑦ ᖃᐅᔨᓴᕈᑎᒃᑲᓐᓂᕐᓂᒃ, ᐋᖅᑭᖑᐊᖅᓯᒪᔪᑦ, ᐋᖅᑭᖑᐊᖅᓯᒪᔪᓂᒃ-ᑐᙵᕕᑖᑦ ᓱᒃᑲᓕᕝᕙᓪᓚᐊᔪᑦ) ᐊᑐᖅᑕᐅᓂᐊᖅᑐᑦ ᐊᕐᕌᒍᓂᐊᖅᑐᓂ, ᑭᓛᓐᕗᐊᑕ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓂᖀᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd) ᐊᑐᓕᖁᔨᔾᔪᑎᓕᐅᖅᓯᒪᔪᖅ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐋᖅᑭᒃᓯᒪᔪᑦ ᐊᑕᐅᓯᕐᒧᑦ ᐊᕐᕌᒍᒧᑦ (2025) ᑕᕝᕙᓂᑐᐊᖅ.

- NC-ᑯᑦ ᑐᓂᓯᓯᒪᔪᑦ ᐅᑯᓂᖓ ᑕᐅᑐᒃᑕᖏᓐᓂᒃ:
  - ᓅᒃᑖᖅ Hᐃᐊᔾᔮᔅ ᐅᓂᒃᑳᖅᑐᖅ ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᑏᑦ ᐊᑐᐃᓐᓇᕈᐃᓯᒪᔪᑦ ᐱᕐᔪᐊᖑᔪᓂᒃ ᐃᓯᒪᒋᔭᐅᔪᓂᒃ, ᐱᓗᐊᖅᑐᒥᒃ ᑐᑭᓕᐅᕆᑎᓪᓗᒋᑦ ᐅᓗᕆᐊᓇᕈᓐᓇᖅᑐᓂᒃ ᕿᒥᕐᕈᓂᕐᓂᒃ ᓴᖅᑭᑕᐅᓯᒪᔪᑦ ᑎᑎᕋᖅᓯᒪᔪᖅ 1.3-ᒥ. ᑖᓐᓇ ᓴᖅᑮᓯᒪᔪᖅ ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᖃᕐᓂᖓ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙶᖅᑐᑦ ᐅᖓᑕᐅᔾᔮᓯᒪᑐᐃᓐᓇᕆᐊᖃᕐᓂᖏᑦ B_{msy}-ᒥᒃ 80%-ᐸᓅᔪᖅ ᐋᖅᑭᖑᐊᖅᑕᐅᓯᒪᔪᓗᒃᑖᓂ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂ, ᑕᒪᓐᓇ ᐅᓗᕆᐊᓇᕈᓐᓇᕐᓂᖓ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙶᖅᑐᑦ ᑐᙶᓄᐊᕐᓗᓂ B_{lim} ᑐᙶᓃᓪᓗᓂ 1% ᐋᖅᑭᖑᐊᖅᑕᐅᓯᒪᔪᓗᒃᑖᓂ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂ ᐊᒻᒪ ᑕᒪᓐᓇ ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᖃᕐᓂᖓ ᐃᖃᓗᒐᓯᖕᓂᖅ ᓖᓇ ᑲᑎᖢᖏᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐅᖓᑕᐅᔾᔮᓗᓂ F_{lim}-ᒥᒃ 34%-ᒥᒃ, ᐅᖓᑖᓃᕋᔭᒃᑐᖅ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᓄᑦ (PA) ᖃᐅᔨᒪᔾᔪᑕᐅᔪᓄᑦ, ᑭᓱᐊᓂᓕ ᑐᙵᕕᒋᓪᓗᖏᑦ ᐅᓄᙱᑦᓛᑦ ᐊᒻᒪᓗ ᒥᒃᓴᐅᓴᒃᑕᐅᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᕈᑎᓂᑦ, ᐊᑐᖅᑕᐅᓂᖓ ᓄᑖᖅ, ᓇᓗᓇᐃᖅᑕᐅᓯᒪᙱᑦᑐᖅ ᓯᓕᔫᓂᖓᓂᒃ ᐊᒻᒪᓗ ᑎᑎᖅᑲᖁᑎᑦᑎᐊᕙᐅᓗᐊᙱᑦᑐᑦ ᕿᒥᕐᕈᓂᕐᒧᑦ ᐋᖅᑭᖑᐊᖅᑕᐅᓯᒪᔪᑦ ᐱᓕᕆᐊᖑᕐᔪᐊᖅᓯᒪᔪᑦ ᖁᓯᙱᓐᓂᕐᒧᑦ ᐊᑯᓚᐃᓕᕇᓂ. ᐅᖃᐅᔾᔨᒋᐊᕈᑏᑦ ᑖᒃᑯᓇᙵᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓂᑦ (SC) ᑐᙵᕕᖃᖅᑰᔨᔪᑦ ᑭᓯᒥᑭᐊᖅ ᐊᓯᐊᓂᒃ ᖃᐅᔨᒪᔨᑕᓪᓚᑦᑖᑦ ᐃᓯᒪᒋᔭᖏᓐᓂᒃ ᐅᕙᓘᓐᓃᑦ ᐃᓯᓕᔾᔪᑕᐅᔪᑦ ᕿᒥᕐᕈᓃᑦ ᐊᒻᒪᓗ ᒪᓕᖕᓃᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᓂᒃ (PA) ᐋᖅᑭᖑᐊᖅᓯᒪᔪᑦ ᐊᑑᑎᖃᕐᓂᖏᓐᓄᑦ ᒥᑭᑦᑐᓪᓚᑯᓗᖕᒥᒃ ᐊᒃᐸᓯᒃᑐᓪᓗᑎᒃ ᖁᓯᙱᑦᑎᐊᕐᓂᕐᓄᑦ. ᑕᒪᓐᓇ ᓈᓪᒪᒃᑑᓕᖅᑎᑦᑎᓯᒪᙱᑦᑐᖅ ᓴᓇᕝᕕᐅᔪᖅ ᐊᑐᙲᓕᕐᓗᑎᒃ ᐋᒃᑲᓐᓂᐊᒍᑦ $20-ᒥᑦ 30-ᒧᑦ ᒥᓕᐊᓐᓂᒃ ᐱᔭᐅᔭᕆᐊᖃᓪᓚᕆᒃᑐᓂᒃ ᑮᓇᐅᔭᓂᒃ ᐃᓯᖅᑐᒃᓴᓂ.
  - ᑐᕌᒐᖅᑲᕐᓃᑦ ᐱᔭᐅᓯᒪᔪᑦ ᑖᒃᑯᓇᙵᑦ ᐋᖅᑭᖑᐊᖅᑕᐅᓯᒪᔪᓂᑦ ᓴᖅᑮᔭᖅᑎᑦᑎᔪᑦ ᓇᓗᓇᓗᐊᔮᕐᓂᖏᑦ ᐊᒻᒪ ᓴᖅᑮᔭᖅᑎᑦᑎᒃᑲᓐᓂᐅᔭᕐᓂᖏᓐᓂᒃ ᖃᓄᐃᓪᒪᑦ ᑰᑕᑦ ᐅᓄᕐᓂᖅᓴᐊᖅᑎᑦᑎᓂᖏᑦ ᐱᔪᓐᓇᖅᑎᑕᐅᙱᓐᓂᖏᓐᓄᑦ. ᐱᔭᐅᓯᒪᔪᖅ ᑖᒃᑯᓇᙵᑦ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ (SC) ᐅᖃᐅᔾᔨᒋᐊᖅᑎᖓᑦ ᒪᒃᐱᒐᖅ 70-ᒥ ᐅᖃᖅᓯᒪᖕᒪᑦ ᐃᒪᓐᓇ: “ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙶᖅᑐᑦ ᓂᕐᔪᑎᓂᙶᖅᑐᓄᑦ ᑐᕌᒐᐅᔪᑦ, ᐊᑕᖏᖅᖢᒋᑦ ᖃᓄᐃᓕᖓᓂᙶᕐᒧᐊᓂ 2025-2026–ᒧᑦ ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙶᖅᑐᑦ ᓂᕐᔪᑎᓂᙶᖅᑐᑦ ᑐᙶᓄᐊᕋᔭᕐᓂᖏᑦ B_{lim} ᑐᙶᓂᓚᐅᖅᑐᖅ 1%. ᑕᐃᒪᐃᓐᓇᔭᕐᓂᖓ ᑕᒪᒃᑯᐊ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙶᖅᑐᑦ ᓂᕐᔪᑎᓂᙶᖅᑐᑦ 2026–ᒥ ᐅᓄᕐᓂᖅᓴᐅᔪᑦ ᐊᑐᒃᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᓄᓇᒥᙶᖅᑐᑦ ᓂᕐᔪᑎᓂᙶᖅᑐᑦ 2024–ᒥ ᐊᑯᓐᓂᖓᓄᑦ 19 ᐊᒻᒪᓗ 70%–ᖑᓪᓗᓂ ᐊᑕᖏᖅᖢᒋᑦ ᑐᕌᒐᐅᔪᓄᑦ.” ᐅᖃᒃᑲᓐᓂᕐᓗᓂ, ᑕᒪᓐᓇ ᓴᖅᑮᔭᖅᑎᑦᑎᔪᖅ ᑰᑕᑦ ᐅᓄᕐᓂᖅᓴᐅᓂᐊᖅᑕᐅᓂᖓ ᓈᓪᒪᒃᑑᓕᖅᑎᑦᑎᓯᒪᙱᑦᑐᖅ.
  - ᑎᓯᐱᕆ 2022-ᒥᓂᑦ, ᐳᓘ ᒫᑐ (Blue Matter) ᐃᖃᓗᓇᓂᕐᓂᑦᒧ ᑐᓴᕋᓯᐊᖅᑎᑦ ᑎᒥᖓᑦ,

ᐱᓕᕆᓯᒪᐃᓐᓇᖅᑐᑦ ᓴᖅᑮᓂᕐᒥᒃ ᐃᓚᑰᔾᔪᑕᐅᔪᓂᒃ ᖃᐅᔨᓴᕐᓂᕐᒥ ᐊᑑᑎᖃᖅᑐᓂᒃ 2018-ᒧᖅ, 2020-ᒧᑦ ᐊᒻᒪᓗ 2021-ᒧᑦ. ᐅᒡᒍᐊᕐᓇᖅᑑᒐᓗᐊᖅ, ᑕᒪᓐᓇ ᐱᓕᕆᓂᖅ ᐱᔭᕇᖅᑕᐅᓯᒪᙱᑦᑐᖅ ᐱᔾᔪᑕᐅᓪᓗᑎᒃ ᑳᓐᓛᓄᑦ/ᑮᓇᐅᔭᖅᑑᑎᓄᑦ ᐱᓇᐃᓗᑕᑦ; ᑭᓯᐊᓂᓕ, ᑎᑎᖅᑲᖁᑎᓂᒃ ᐊᒃᑐᐃᑐᐃᓐᓇᕆᐊᓕᒃ ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᕿᒥᕐᕈᓂᕐᒥ.

- NC-ᑯᑦ ᐅᒃᐱᕈᓱᒃᑐᑦ ᑖᒃᑯᐊ ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ (SC) ᐊᑐᓕᖁᔨᔾᔪᑎᖏᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐅᓄᕈᓐᓃᕆᐊᖅᑎᑕᐅᓂᖓ ᓈᒻᒪᓚᐅᙱᑦᑐᖅ ᐊᒻᒪᓗ ᓈᒻᒪᒃᑑᓕᖅᑎᑦᑎᓯᒪᙱᑦᑐᖅ, ᑐᙵᕕᒋᓪᓗᒋᑦ ᓈᒻᒪᙱᑦᑐᑦ ᑐᑭᓕᐅᕆᓃᑦ ᑕᕝᕙᓂ ᐅᔾᔨᕐᓱᖅᓂᒃᑯᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᓂᒃ (PA). ᐅᐊᖕᓇᒥᒃ ᐊᑦᓛᓐᑎᒃᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ (NAFO) ᑭᒡᓕᓯᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔩᑦ (SC) ᑐᓴᐅᒪᔾᔪᑎᖃᓚᐅᙱᑦᑐᑦ ᕿᒥᕐᕈᓂᕐᒧᑦ ᐱᑕᖃᕐᕕᐅᔪᓂ ᐊᒻᒪ ᐃᓕᓯᓯᒪᓪᓗᑎᒃ ᐃᓚᑰᔾᔪᑕᐅᔪᓂᒃ ᖃᐅᔨᓴᕈᑎᓂᒃ ᐃᑲᔫᑎᖃᖅᓯᒪᔪᑦ ᐅᓗᕆᐊᓇᕈᓐᓇᕐᓂᖏᓐᓄᑦ ᐱᑕᖃᕐᕕᐅᔪᑦ. ᐊᑐᓕᖅᑎᑦᑎᓂᖅ ᑕᒪᑐᒥᖓ ᓄᑖᒥᒃ ᐅᓄᕐᓂᖏᓐᓄᑦ ᐋᖅᑭᙳᐊᖅᑕᐅᓯᒪᔪᓄᑦ ᑭᒡᓕᖃᖅᑎᑐᓂᒃ ᐊᒻᒪ ᐊᔪᙱᓪᓚᕆᒃᑐᓂᖒᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᓇᓚᐅᖅᑖᖅᑕᐅᓯᒪᔪᑦ, ᐱᑕᖃᙱᓐᓂᖏᑦ ᖃᓄᑐᐃᓐᓇᖅ ᐊᕙᑎᒥᒃ ᐊᒻᒪᓗ ᐆᒪᓂᖃᖃᑎᒌᑦ ᑎᑎᖅᑲᖁᑎᖏᓐᓂᒃ, ᐊᔾᔨᒌᙱᕈᓘᓴᖅᑐᓪᓗ ᖁᓯᙱᓐᓂᕐᒧᑦ ᐊᑯᓚᐃᓕᓈᑦ ᐅᒃᐱᕐᓇᓗᐊᙱᒻᒪᑕ ᑕᒪᑐᒧᖓ ᑰᑕᑦ ᐅᓄᕈᓐᓃᕆᐊᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐊᑐᓵᖅᑐᓂ ᒪᕐᕉᖕᓂ ᐊᕐᕌᒍᖕᓂ. ᓈᒻᒪᒃᑐᙳᖅᑎᑦᑎᓯᒪᙱᓪᓚᕆᒃᑐᑦ ᐅᓄᕈᓐᓃᕆᐊᓪᓚᒃᑎᑕᐅᓂᖏᓐᓄᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ. ᑕᐃᒪᐃᓂᖓᓄᑦ, NC-ᑯᑦ ᐊᑐᓕᖁᔨᔪᑦ ᐊᑐᖅᑕᐅᒃᑲᓐᓂᓕᕐᓗᑎᒃ 2022-ᒥ ᑰᑕᑦ 36,400-ᖑᔪᑦ ᑕᓐᓯᑦ 2025-ᒧᑦ.
- ᒫᓐᓇᓕᓴᑦ ᖃᐅᔨᓴᕐᓂᕐᒥ ᖃᐅᔨᔭᐅᔪᑦ 2022-ᒥ ᐊᒻᒪᓗ 2023-ᒥ ᐱᐅᔫᔪᑦ. ᐊᒃᑐᐃᓗᐊᖅᑕᐃᓕᒪᖁᓪᓗᒋᑦ ᐅᓗᕆᐊᓇᓗᐊᙱᑦᑐᓄᑦ ᐱᑕᖃᕐᕕᐅᔪᑦ ᓄᖑᓕᕐᓂᖏᓐᓄᑦ ᒫᓐᓇ ᐃᖃᓗᒐᓱᓐᓃᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ, ᐅᖃᐅᔾᔨᒋᐊᕈᑎᑦᑎᐊᕙᒃ ᕿᒥᕐᕈᔭᐅᓗᑎᒃ 2024-ᒥ ᐅᑭᐅᒃᓵᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᕐᒥ ᖃᐅᔨᔭᐅᔪᑦ, ᑲᑎᙵᓪᓗᑎᒃ ᖃᐅᔨᓴᕈᑎᒃᓴᓐᓂᕐᓄᑦ ᐊᑐᐃᑐᓐᓇᐅᑐᐃᓐᓇᕆᐊᓖᑦ (ᓲᕐᓗ ᕿᒥᕐᕈᔭᐅᓃᑦ ᐳᓘ ᒫᑐᑯᓐᓂᑦ [Blue Matter]) ᐋᖅᑭᒃᑕᐅᖅᑳᖅᑎᓐᓇᒋᑦ ᑲᑎᖢᒋᑦ ᐊᖑᓂᒃᑕᐅᔪᓐᓇᖅᑐᑦ 2026-ᒥ.

ᖃᓄᐃᓕᐅᕈᑎ:

- ᑲᑎᒪᓂᕐᒧᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ ᐱᑕᖃᖅᑎᑕᐅᓂᐊᖅᑐᑦ.
- ᑎᑎᕋᖅᓯᒪᔪᑦ ᑐᓂᔭᐅᔪᑦ ᑖᒃᑯᓇᙵᑦ ᑲᓇᖕᓇᖓᓂ ᐃᖅᑲᒥᐅᑕᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔭᑕᕆᔭᐅᔪᑦ ᑲᑎᒪᔨᕋᓛᓂᑦ (EAGSAC) ᐃᓚᒋᔭᐅᔪᓂᑦ ᑐᓂᔭᐅᓯᒪᔭᒋᐊᓖᑦ ᓴᓇᕝᕕᐅᔪᒥᑦ ᐅᓪᓗᑐᐃᓐᓇᖅ, ᐋᒡᒋᓯ 23, 2024-ᖑᓚᐅᖅᑳᖅᑎᓐᓇᒍ.

ᑐᓂᓯᐅᔪᑦ ᐅᑯᓇᖓ
ᓄᓇᕗᑦᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ

ᐆᒧᖓ

ᑐᑭᓯᒋᐊᒃᑲᓐᓂᓂᕐᒧᑦ: X ᐃᓱᒪᓕᐅᓂᕐᒧᑦ: ᐊᑐᓕᕐᑯᔭᐅᔪᑦ:

ᐱᔾᔪᑎᖓ: ᑐᑭᓯᒋᐊᕈᑏᑦ ᐱᕙᓪᓕᐊᑎᑦᑎᖃᑎᒌᓐᓂᕐᒧᑦ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓖᑦ

ᖃᓄᐃᓕᐅᕐᓂᐅᓯᒪᔪᑦ:

ᐃᖃᓗᓕᕆᓂᐅᔪᑦ ᓄᓇᕗᑦ ᓇᔪᖅᑕᖓᓂ (NSA) ᒫᓐᓇᐅᔪᒥᑦ ᐊᐅᓚᑕᐅᔪᑦ ᐊᑖᓂ ᒫᓐᓇ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᓂᖃᖕᒋᑦᑐᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᑐᓕᖅᑎᑎᖕᒋᑦᑐᑦ ᐃᓄᐃᑦ ᐱᔪᓐᓇᐅᑎᖕᒋᓐᓂ ᐊᒻᒪ ᐊᔾᔨᐅᖕᒋᑦᑐᑦ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᒥᐅᔪᓂᑦ ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᔪᓂᑦ *ᓄᓇᕗᒻᒧᑦ ᐊᑭᒻᒥᑦ* (*NA*). ᓄᑖᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓖᑦ ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᔪᑦ ᐱᑕᖃᖅᑎᑦᑎᑐᑎ ᐊᔾᔨᒌᓐᓂᖃᖅᑐᓂ ᐊᖑᓇᓱᓐᓂᕐᒧᑦ ᐱᔪᓐᓇᐅᑎᓂᑦ ᐊᒻᒪ ᐆᒪᔪᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᑦ ᓴᖅᑭᖅᑎᑕᐅᓯᒪᔪᑦ ᐊᑖᓂ *ᓄᓇᕗᒻᒧᑦ ᐊᑭᒻᒥᑦ*, ᐱᕕᔅᓴᖅᑎᑦᑎᑐᑎ ᓯᕗᒧᒋᐊᖅᑎᑕᐅᓂᖓᓄᑦ ᑲᓇᒧ ᐋᖅᑭᒍᑎᖃᓂᕐᒧᑦ ᐱᓇᓱᐊᖅᑕᐅᔪᓂᑦ ᓄᓇᖃᖅᑳᖅᓯᒪᔪᓄᑦ ᓄᓇᓕᓐᓄᑦ, ᐃᑲᔪᕐᓱᒍ ᐃᓄᐃᑦ ᓄᓇᖓᑦ ᐊᑐᐊᒐᖅ ᓯᕗᒧᑦᑎᒋᐊᕐᓗᒋᑦ ᐃᓄᐃᑦ ᓇᒻᒥᓂᖅ ᐱᔪᒪᓂᖃᓂᕐᒧᑦ, ᐊᒻᒪ ᓴᖅᑭᑎᑦᑎᓗᑎ ᐅᓪᓗᒥᒨᖓᔪᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᒥᐅᔪᒥᑦ ᓇᓗᓇᐃᕐᓗᒍ ᒪᑭᒪᑎᑕᔅᓴᐅᓂᖏᑦ ᐃᖃᓗᓕᕆᓂᐅᔪᑦ ᓄᓇᕗᑦ ᓇᔪᖅᑕᖓᓂ.

ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᐊᐱᕆᔪᐅᓚᐅᖅᑐᑦ ᐱᑕᖃᖅᑎᑦᑎᑐᑎ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖓᓄ ᑐᔅᓯᕌᒧᑦ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ (NFR). ᑕᒃᑯᐊ ᑐᑭᓯᒋᐊᕈᑏᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᑐᕌᖓᓇᓱᐊᖅᑐᑎ ᓇᑦᑐᒥ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᑐᑎ ᐱᕙᓪᓕᐊᑎᑦᑎᖃᑎᒌᓐᓂᕐᒧᑦ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ, ᐊᒻᒪᓗᑦᑕᐅᖅ ᑐᑭᓯᒋᐊᕈᑎᔅᓴᓐᓂᕐᓂᑦ ᐃᓚᖏᓐᓂ ᐃᓗᓕᓪᓗᐊᑕᖏᓐᓂ ᑐᔅᓯᕌᖑᔪᒥᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ.

ᐱᑯᓯᑐᖃᖓ

ᐅᕙᓂ ᔪᓚᐃ 9, 2018, ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ (NTI), ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ (NWMB), ᐊᒻᒪ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ (GN) ᓴᖅᑭᑎᑦᑎᓚᐅᖅᑐᑦ ᑲᑐᑎᔭᐅᔪᒥ ᐅᓂᒃᑳᓂᑦ ᑲᔪᖏᖅᓯᒪᒃᑲᓂᓂᕐᒧᑦ ᓴᖅᑭᑎᑦᑎᖃᑎᒌᓐᓂᕐᒧᑦ ᓄᑖᓂ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ ᓄᓇᕗᑦ ᓇᔪᖅᑕᖓᓄ. ᑲᑐᑎᔭᐅᔪᒥ ᐅᓂᒃᑳ ᓴᖅᑭᖅᑎᑕᐅᔪᒻᒪᓂᑦ, ᒪᑭᕕᒃ ᑯᐊᐳᕇᓴᓐ (ᒪᑭᕕᒃ) ᐅᓚᒋᔭᐅᓕᓚᐅᖅᑐᑦ ᐅᖃᖃᑎᒌᓐᓂᐅᔪᓂᑦ ᓇᒦᓐᓂᖏᑦ ᐊᔾᔨᖓᓂ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᓇᔪᖅᑕᐅᔪᓂᑦ. ᑲᑎᒪᔨᕋᓛᑦ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᓴᖅᑭᑎᑦᑎᖃᑎᒌᑦᑐᑎ ᐊᑐᐊᒐᕐᓄᑦ ᐃᓂᒋᔭᖓᓂ ᖃᐅᔨᑎᑦᑎᓂᐊᖅᑐᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖏᓐᓄᑦ ᓄᑖᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓖᑦ. ᑲᑎᒪᔨᕋᓛᓂᖅᑕᐅᔪᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ, ᒪᓕᒐᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ, ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ, ᒪᑭᕕᒃ, ᐊᒻᒪ ᐅᓇᐅᓕᖅᑎᓪᓗᒍ 2023, ᓄᓇᕗᑦᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ (NMRWB).

ᓄᑖᑦ ᐊᐅᓚᓂᖏᑦ ᐊᓕᖅᑎᑕᐅᓂᖓᓄᑦ ᑲᑎᒪᔩᑦ–ᒥᓂᔅᑕᒧ ᐃᓱᒪᓕᐅᕐᓂᐅᔪᑦ

ᐊᑖᓂ ᐃᓚᖓ 5 *ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᑦ*, ᒥᓂᔅᑕ ᐱᓕᕆᐊᖃᕆᐊᖃᖅᑐᖅ ᐊᑐᓕᖅᑎᑦᑎᓗᓂ ᑭᖑᓪᓕᖅᐹᒥ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᓂᑦ ᓴᖅᑭᖅᑐᓂᑦ ᑲᑎᒪᔩᓂ/ᒥᓂᔅᑕᒥ ᐱᕙᓪᓕᐊᓂᖓ. ᓄᓇᕗᒻᒥ ᓇᔪᖅᑕᖓᓂ, ᑐᓐᖓᓂᖅᑐᑦ ᐊᒥᓱᓚᖓᓄ ᐱᔪᓐᓇᖅᑎᑕᐅᔪᓂᑦ, ᐃᓄᐃᑦ ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᖏᑦᑐᑦ ᓚᐃᓴᓐᓯᓂᑦ ᐊᖑᓇᓱᓐᓂᕐᒧᑦ ᐱᓕᕆᔭᐅᔪᓂᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᑐᙵᕕᖃᓂᖅ ᓚᐃᓴᓐᓯᓂᑦ ᐊᒻᒪ ᖃᓄᐃᓐᓂᓇᓕᕆᐊᖃᖅᑕᖅᑎᖏᓐᓂ ᐱᔪᓐᓈᖏᑦᑐᑦ ᐊᑐᓕᖅᑎᑦᑐᒍ ᑲᑎᒪᔩᓄᑦ/ᒥᓂᔅᑕᒧᑦ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᑦ ᐊᒥᓱᓂᑦ ᖃᓄᐃᓕᖓᓂᐅᔪᓂᑦ, ᐊᒻᒪ ᒫᓇᐅᔪᒥᑦ ᐊᓯᐊᒍᑦ ᒪᓕᒐᖅᑎᒍᑦ ᐊᖅᑯᑎᑕᖃᖅᑐᖅ ᐊᑐᓕᖅᑎᓪᓗᒋᑦ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᑦ ᐱᑕᖃᕆᐊᖃᖅᑎᑦᑎᔪᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂ ᐊᖑᓇᓱᑦᑎᓄᑦ.

ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ (NADIO) ᑐᔅᓯᕌᖑᓯᒪᔪᖅ ᐋᖅᑭᒍᑎᐅᓗᓂ ᑕᔅᓱᒧᖓ ᐱᔾᔪᑕᐅᔪᒧᑦ ᐊᒻᒪ ᐊᐅᓚᓂᖃᓂᐊᖅᑐᖅ ᒪᓕᒐᖅᑎᒍᑦ ᓴᓇᕐᕈᑎᐅᓗᓂ ᐱᔪᓐᓇᖅᑎᑕᐅᓗᓂ ᐱᓕᕆᕝᕕᐅᔪᖅ ᐊᑐᓕᖅᑎᓪᓗᒋᑦ ᑲᑎᒪᔨᓄᑦ/ᒥᓂᔅᑕᒧᑦ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᑦ. ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ ᓇᓗᓇᐃᖅᓯᓂᐊᖅᑐᖅ ᒥᓂᔅᑕ ᐊᑐᕈᓐᓇᖅᑐᓂᒋ ᐱᓕᕆᐊᒋᒋᐊᓕᖏᑦ ᐊᑖᓂ *ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᖁᔭᖅ* ᐊᒻᒪ *ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᑦ*.

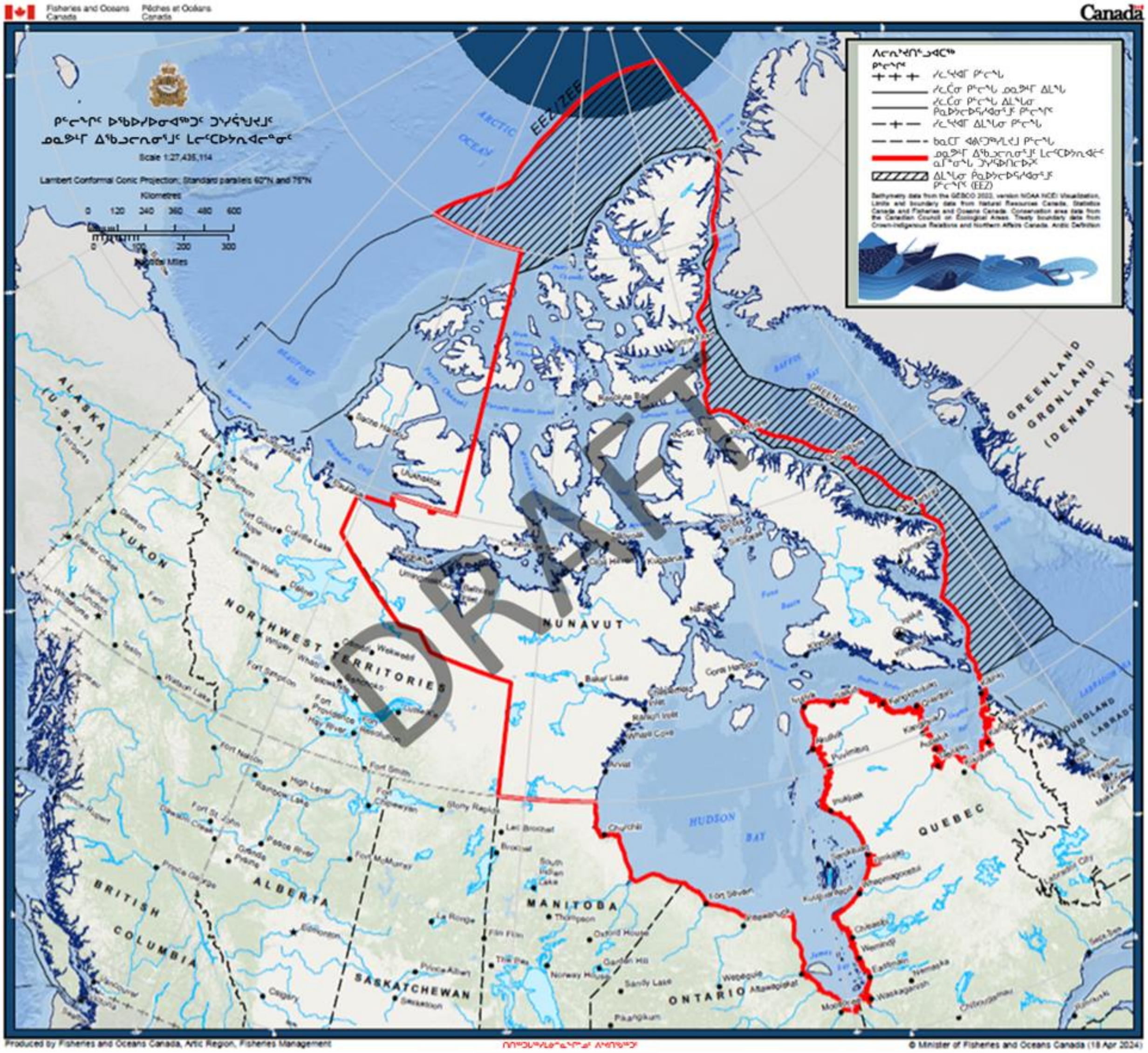
ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ ᑐᔅᓯᕌᖑᔪᖅ ᐊᑐᖅᑎᑕᐅᓗᓂ ᐃᓗᐊᓂ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ ᑐᔅᓯᕋᐅᑎᓕᐅᓂᕐᒧᑦ (ᑕᑯᓗᒍ ᐊᔾᔨᖑᐊᖅ 1) ᐃᓗᐊᓃᑦᑐᖅ ᓄᓇᕗᑦ ᓇᔪᖅᑕᖓᓂ, ᐊᒻᒪ ᐱᖃᓯᐅᔾᔨᓂᐊᖅᑐᑦ ᐊᔾᔨᒌᖏᑦᑐᓂ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᓕᕆᔭᐅᔪᓂᑦ ᓇᓗᓇᐃᔭᖅᑕᐅᓯᒪᔪᑦ *ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᒻᒥᑦ*:

- ᐊᒥᓲᓂᖏᓐᓄᑦ ᐃᖃᓗᑦᑕᐅᔪᓐᓇᖅᑐᑦ ᑭᓪᓕᖏᑦ ᓲᕐᓗ ᑲᑎᑦᑐᒋᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ (TAH);
- ᑰᑕᖃᖏᑦᑐᓄ ᑭᓪᓕᖃᖅᑎᑕᐅᓂᖏᑦ ᑐᑭᖃᖅᑐᖅ ᖃᓄᐃᑦᑐᑐᐃᓐᓇᕐᓂ, ᑭᓯᐊᓂ ᑲᑎᑦᑐᒋᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ, ᐊᒻᒪ ᐱᖃᓯᐅᔾᔨᔪᓐᓇᖅᑐᑦ ᑭᓪᓕᖃᖅᑎᑕᐅᓂᖏᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐊᖑᓇᓱᓐᓂᕐᒧᑦ, ᐊᖑᑕᐅᓂᖓ ᐊᕐᓇᐅᓂᖓ ᐆᒪᔪᐃᑦ, ᐊᖏᓂᖏᑦ ᐆᒪᔪᐃᑦ, ᐊᕐᕌᒍᖏᑦ ᐆᒪᔪᐃᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᐊᑐᖅᑕᐅᔪᑦ ᐊᖑᓇᓱᓐᓂᕐᒧᑦ; ᐊᒻᒪ,
- ᖃᐅᔨᓴᖃᑦᑕᓂᕐᒧᑦ ᐊᒻᒪ ᑐᓴᖅᑎᑦᑎᓂᕐᒧᑦ ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᔪᑦ.

ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓖᑦ ᓇᓗᓇᐃᖅᓯᓂᐊᖅᑐᑦ ᑖᒃᑯᐊ ᐃᓄᓕᒫᑦ ᑎᓕᓯᔾᔪᑎᒧᑦ ᐊᑐᖅᑎᑕᐅᔪᑦ ᒪᓕᒋᐊᖃᖅᑐᑦ, ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ ᐊᔭᐅᕋᔅᓴᐅᑎᓪᓗᒍ ᐃᖃᓗᓕᕆᔨᓄᑦ ᐊᑖᓂ *ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᖁᔭᖅ*. ᐅᑯᓇᓂ ᑐᔅᓯᕋᐅᑎᓕᐅᓂᕐᒧᑦ, ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ ᑐᓂᔭᐅᒐᔭᖅᑐᖅ ᑭᓯᐊᓂ ᑭᖑᓂᐊᒍᑦ ᑭᖑᓪᓕᖅᐹᒥ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᓂᑦ ᑲᑎᒪᔨᓄᑦ/ᒥᓂᔅᑕᒧᑦ ᐱᕙᓪᓕᐊᓂᖃᖅᓯᒪᑎᓪᓗᒋᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᐊᑐᓕᖅᑎᓪᓗᒍ ᓖᓐᓇᐅᔪᒥᑦ ᐃᓱᓕᐅᖅᑕᐅᔪᑦ ᐱᓕᕆᐊᖃᕐᓗᑎ ‘ᑐᐊᕕᕐᓇᖅᑐᓂ ᐊᒻᒪ ᐊᔾᔨᐅᖏᑦᑐᓂ’ ᖃᓄᐃᓕᖓᓂᐅᔪᓂᑦ.

ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ ᐃᓂᖓᓃᖏᓐᓇᓂᐊᖅᑐᖅ ᒪᓕᓐᓂᖃᕐᓗᓂ ᓇᒻᒥᓂᖅ ᐅᖃᖅᓯᒪᓂᖏᓐᓄᑦ (ᓲᕐᓗ, ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᓂᑦ ᑭᖑᓪᓕᖅᐹᒥ ᐃᓱᒪᓕᐅᖅᑕᐅᔪᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᑭᓯᐊᓂ ᐊᓯᔾᔨᖅᑕᐅᒍᓂ ᑲᑎᒪᔨᓄᑦ/ᒥᓂᔅᑕᒧᑦ ᐱᕙᓪᓕᐊᓂᖓ); ᐱᑕᖃᖏᑦᑐᖅ ᓯᕗᓂᐊᒍᑦ ᐋᖅᑭᔅᓯᒪᔪᓂᑦ ᐊᑯᓂᐅᓛᓇᔪᓐᓇᖅᑕᖓ ᑭᓪᓕᖓ ᐊᑯᓂᐅᓂᖓᓄᑦ ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎ. ᐋᖅᑭᒋᐊᖅᑕᐅᓂᖏᑦ ᐅᑭᐅᑕᖅᑐᒥ ᐊᖏᕈᑎᓄᑦ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐊᑐᓕᖅᑎᐅᓂᖓᓄ ᑎᓕᓯᔾᔪᑎᒧᑦ ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᒍᓂ, ᑕᐃᒪᐃᑕᐅᒋᐊᖃᕋᔭᖅᑐᖅ ᐅᑭᑎᒎᓇ ᑲᑎᒪᔩᑦ/ᒥᓂᔅᑕ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐱᕙᓪᓕᐊᓂᖓᒍᑦ.

ᓇᓗᓇᐃᕈᑎ ᑐᔅᓯᕋᖅᓯᒪᖏᑦᑐᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᑲᑎᒪᔩᑦ/ᒥᓂᔅᑕ ᐃᓱᒪᓕᐅᓂᕐᒧᑦ ᐱᕙᓪᓕᐊᓂᖓ ᓇᓗᓇᐃᔭᖅᑕᐅᓯᒪᔪᑦ ᐅᕙᓂ ᐃᓚᖓ 5 *ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᒻᒥᑦ* ᐃᓚᒋᓪᓗᓂᐅᒃ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓖᑦ ᐅᖃᖃᑎᒌᓐᓂᐅᔪᑦ. ᐃᖃᓗᓕᕆᓂᐅᔪᑦ ᓄᓇᕗᑦ ᓇᔪᖅᑕᖓᓂ ᑲᔪᓯᓂᐊᖅᑐᖅ ᐊᐅᓚᑦᑎᖃᑎᒌᓐᓂᐅᔪᓄᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᒻᒪ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ.



ᐊᔾᔨᙳᐊᖅ 1: ᓄᓇᖑᐊᖅ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᖅ ᑐᓯᕋᖑᔪᒥᑦ ᐊᑕᖏᖅᑐᒍ ᓄᓇᖕᖓᓄ ᓇᒦᓐᓂᖕᖓᓄ ᑐᓯᕋᐅᑎᓕᐅᕈᑦ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᒪᓕᑦᑕᐅᓯᕆᐊᓕᓐᓂᑦ (ᐊᐅᐸᖅᑐᖅ ᑎᑎᕋᖅᓯᒪᔪᖅ).

ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ

ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ (CFP) ᑐᓯᕋᖑᔪᑦ ᓄᑖᖑᓗᑎ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐅᓚᑦᑎᓂᕐᒧᑦ ᓴᓇᕿᑎᐅᓗᓂ. ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ ᓴᖅᑭᖅᑎᑕᐅᓚᔭᖅᑐᑦ ᐃᓄᐃᑦ ᐆᒪᔪᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒌᓄᑦ (IWMOs) ᐊᐅᓚᓗᒋ ᐃᓄᐃᑦ ᐊᖑᓇᓱᓂᕐᒧᑦ ᐃᖃᓗᓐᓂᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐃᒪᕐᒥᐅᑕᐃᑦ ᐆᒪᔪᐃᑦ, ᐊᒻᒪ ᐱᖃᓯᐅᔾᔨᒍᓐᓇᖅᑐᑦ ᐊᖑᓇᓱᓂᒧᑦ ᑭᓪᓕᖅᑲᖅᑎᑕᐅᔪᓂᑦ. ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ ᐊᑐᕋᔭᖅᑐᖅ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐃᓗᐊᓂ ᓄᓇᕘᑦ ᓇᔪᖅᑕᖕᖓᓂ ᐊᒻᒪ ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᓚᔭᖅᑐᖅ ᐃᖃᓗᓐᓄᑦ ᐅᕙᓗᓐᓃᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᑖᒃᑯᐊ ᐃᓄᐃᑦ ᐊᖑᓇᓯᖃᑦᑕᖅᑕᖏᑦ "ᐊᖏᔫᓪᓗᑎ ᓂᐅᕕᐊᕐᓴᐃᑦ", ᓅᕐᓗ ᓂᐅᕕᐊᕐᓴᐃᑦ ᐃᖃᓗᓕᕆᕝᕕᒧᑦ, ᐃᖅᑲᓇᐃᔭᓂᕐᒧᑦ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐅᕙᓗᓐᓃᑦ ᓂᕆᕙᓐᓂ (ᐊᒻᒪ ᐱᖃᓯᐅᑎᓐᓇᑎ ᓂᐅᕕᐊᕐᓴᐃᑦ ᑐᕌᖅᑕᐅᑎᒋᔪᓂᑦ ᐃᓄᓐᓄᑦ ᓂᐅᕕᖃᑦᑕᖅᑐᓄᑦ). ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ ᐊᓯᐊᒍᓪᓕ ᓇᓖᕌᕋᔅᓴᐅᓚᔭᖅᑐᖅ ᓴᓇᕿᑎᐅᓗᓂ ᐃᑲᔪᕐᓗᓂ ᓄᓇᓕᒥ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᖃᓗᓕᕆᓂᕐᒥᑦ ᐊᒻᒪ ᐃᓄᐃᑦ ᓇᒻᒥᓂᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᐊᔭᐅᖅᑕᐅᓂᖏᑦ ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ ᑲᒪᒋᔭᕆᐊᖃᕋᔭᖅᑕᖏᑦ ᐃᓄᐃᑦ ᐆᒪᔪᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒋᓄᑦ. ᑭᓯᐊᓂᓕ, ᐃᓄᐃᑦ ᐆᒪᔪᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᑐᔾᔨᖃᑎᒋᑦ ᑐᔅᓯᕋᕈᓐᓇᖅᑐᑦ ᑖᒃᑯᐊ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐃᓚᓯᓗᑎ ᐃᓗᓕᖏᓐᓂ ᓄᓇᓕᒻᒧᑦ ᐃᖃᓗᓐᓂ ᐸᕐᓇᐅᑏᑦ ᐊᔭᐅᕋᔅᓴᐅᔪᓂᑦ ᐊᑖᓂ *ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐱᖁᔭᖅ*. ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐃᓱᒪᖅᓲᑎᖃᕋᔭᖅᑐᑦ ᐊᖏᕐᓗᑎ ᑖᒃᑯᓂᖓ ᑐᔅᓯᕋᐅᑎᓂᑦ.

ᐊᐃᑉᐸᖓᓂ ᐱᓕᕆᒋᐋᖅᓃᑦ:

ᐊᐅᓚᑦᑎᖃᑎᒌᓐᓂᕐᒧᑦ ᐱᓕᕆᖃᑎᖏᑦ ᒫᓐᓇᐅᔪᒥᑦ ᓴᖅᑭᑎᑦᑎᖃᑎᒌᓐᓂᖃᖅᐸᓪᓕᐊᔪᑦ ᐊᑐᐊᒐᕐᓂᑦ ᐱᔾᔪᑎᖏᓐᓄ ᐸᐃᑉᐸᑦ (PIP), ᓇᓗᓇᐃᔭᐃᓂᐊᖅᑐᑦ ᐊᑐᐊᒐᕐᓂ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᕌᒐᖏᓐᓂ ᑐᔅᓵᖑᔪᓂ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ. ᑐᕌᒐᕆᔭᖓ ᑎᑎᕋᕆᐊᓐᖓᖅᑕᐅᓗᑎ ᐊᑐᐊᒐᕐᓂᑦ ᐱᔾᔪᑎᖏᓐᓄ ᐸᐃᑉᐸᑦ ᐅᕙᓂᐅᓕᖅᑎᓪᓗᒍ ᑎᓯᐱᕆ 2024. ᑭᖑᓂᐊᒍᑦ ᑖᔅᓱᒪ, ᐱᔾᔪᑎᒋᓇᓱᐊᖅᑕᖓ ᑐᓂᐅᖅᑲᕐᓗᖏᑦ ᐊᑐᐊᒐᕐᓂᑦ ᐱᔾᔪᑎᖏᓐᓄ ᐸᐃᑉᐸᑦ ᓯᓚᑖᓄᑦ ᖃᐅᔨᑎᓪᓗᒋᑦ ᑲᑎᒪᖃᑎᖃᕐᓂᐅᔪᑦ ᐱᔪᓐᓇᐅᑎᓂᑦ ᑎᒍᒥᐊᖅᑎᓂᑦ, ᑎᒍᒥᐊᖅᑎᐅᖃᑕᐅᔪᓂᑦ, ᐊᒻᒪ ᐊᓯᖏᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᒐᕙᒪᐃᑦ ᐱᒋᐊᕐᓂᖏᓐᓂ 2025, ᐱᖃᓯᐅᑎᓪᓗᑎ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ. ᑎᑎᖅᑲᐃᑦ ᓇᓗᓇᐃᔭᐃᔪᑦ ᑐᔅᓵᖑᔪᒥᑦ ᓇᒦᓐᓂᖓ ᑐᔅᓯᕋᐅᑎᓕᐅᕈᑦ, ᐊᒻᒪ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᑐᓂᓯᓂᐊᖅᓯᒪᓂᖏᓐᓄᑦ ᑎᑎᕋᖅᑕᐅᒋᐊᓐᖓᖅᓯᒪᔪᓂᑦ ᐊᑐᐊᒐᕐᓂᑦ ᐱᔾᔪᑎᖏᓐᓄ ᐸᐃᑉᐸᑦ, ᐊᐅᓪᓚᖅᑎᑕᐅᓯᒪᔪᑦ ᓄᓇᓕᓐᓄᑦ ᐊᒻᒪ ᑲᑐᔾᔨᖃᑎᒌᓄᑦ ᑭᒡᒐᖅᑐᐃᔪᓂᑦ ᐱᔪᓐᓇᐅᑎᓂᑦ ᑎᒍᒥᐊᖅᑎᓂᑦ ᓄᓇᕗᒻᒥ.

ᑐᓂᔾᐅᔪᑦ ᐅᑯᓄᖓ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᓃᖅᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ

ᐅᑦᑑᐱᕆ 2024

ᐅᑯᓄᖓ

ᑐᑭᓯᐊᕈᑏᑦ: X ᐃᓱᒪᓕᐅᓂᕐᒧᑦ:

ᐱᔾᔪᑎᖓ: ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO) ᑐᑭᓯᐊᕈᑏᑦ – ᐃᒪᐃᑦ ᓴᐳᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᖏᓐᓂᖓ

ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ
ᐃᓚᒋᒡᓗᓂᐅᒃ ᐃᒪᐃᑦ ᓴᐳᒥᔭᐅᓂᖏᓐᓄᑦ ᑐᕌᖅᑕᐅᔪᑦ ᐱᓕᕆᔭᐅᔪᑦ, ᑲᓇᑕᐅᑉ ᒐᕙᒪᒃᑯᖏᑦ ᑲᔪᖏᖅᓯᒪᔪᑦ ᓯᕗᓕᕆᐊᖅᑎᑕᐅᓂᖓ ᐱᕚᓪᓕᐊᓂᖓ ᐊᐅᓚᑦᑎᓂᖅᑲᑦᑎᐊᓂᕐᒥᑦ ᐱᑕᖃᓈᖅᑐᓂᑦ ᐃᒪᐃᑦ ᓴᐳᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᖏᓐᓂᖏᑦ (MPAs) ᐊᒻᒪ ᐊᓯᖏᑦ ᓇᖏᓐᓂᖏᑦ ᑐᓐᖓᓂᖃᖅᑐᑦ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᑦ (OECMs). ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐃᖅᑲᓇᐃᔭᖃᑎᖃᐃᓐᓇᖅᑐᑦ ᐱᓕᕆᖃᑎᖏᓐᓂ ᓯᕗᓕᑦᑎᑕᐅᑎᐊᖅᑐᓂ ᓴᖅᑭᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᓄᑖᑦ ᓇᔪᖅᑕᐅᔪᑦ (ᐃᓚᒍᑕᖓ 1) ᑐᓂᓯᓂᖃᕋᔭᖅᑐᑦ ᑲᓇᑕᐅᑉ ᑐᕌᖓᓂ ᓴᐳᒻᒥᑦᑎᓗᑎ 30% ᑲᓇᑕᐅᑉ ᐃᒪᖏᓐᓂ 2025–ᖑᓕᖅᑎᓪᓗᒍ, ᐊᒻᒪ ᑐᕌᕐᓂᖃᖅᑐᑎᑦ ᓯᓚᕐᔪᐊᒥ ᑐᕌᖅᑕᐅᔪᖅ ᓴᐳᒻᒥᑎᓗᑎ 30% 2030–ᖑᓕᖅᑎᓪᓗᒍ.

ᑐᑭᓯᐊᕈᑏᑦ

1. *ᕿᑭᖅᑕᐃᑦ ᐊᒻᒪ ᓴᕐᕙᕐᔪᐊᖅ ᖃᐅᔨᓴᕐᕕᒋᔭᐅᔪᑦ*

- ᓴᕐᕙᕐᔪᐊᖅ (ᑲᓇᑕᒥ ᐃᓚᖓ ᐅᐊᓐᓇᖓᓂ ᐃᒪᖓ ᐃᒪᐅᖏᓐᓇᖅᑐᖅ) ᐊᒻᒪ ᕿᑭᖅᑕᐃᑦ (ᐃᒪᖏᑦ ᖃᓂᒋᔭᖓᓂ ᓴᓂᑭᓗᐊᖅ ᕿᑭᖅᑕᖏᑦ ᓇᒥᓂᖓ) ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓄᑦ (QIA) 2022 ᑎᑎᕋᖅᓯᒪᔪᑦ ᐃᑲᔪᕐᓗᒍ ᐊᕙᑦᑐᖅᓯᒪᔪᒧ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓂᖓ ᐊᒻᒪ ᑐᓂᓯᓂᖃᖅᑐᑎ ᑐᕌᖅᑐᓂ ᓯᕗᓕᕆᐊᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐃᓄᓐᓄᑦ ᑐᑭᒧᐊᖅᑎᑕᐅᔪᑦ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᕿᑭᖅᑖᓗᒻᒥ.
- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ, ᐃᓚᒋᒡᓗᒋᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᒻᒪ ᓯᓚᖓᓄ ᐊᓯᔾᔨᕐᓂᖓᓄ ᑲᓇᑕᒥ (ECCC) ᐊᒻᒪ ᐃᖏᕋᔪᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ,, ᑐᑭᒧᐊᖅᑎᑦᑎᔪᑦ ᒐᕙᒪᓕᒫᒧᑦ ᐋᔩᖃᑎᒌᓐᓂᕐᒥᑦ ᐃᓄᓐᓄᑦ ᐊᑦᑐᖅᑕᐅᔪᓄᑦ ᐊᒻᒪ ᐃᑲᔫᓯᐊᓄᑦ ᐊᖏᕈᑦ (IIBA) ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᒃᑯᓐᓂ ᐃᑲᔪᖅᑐᑎ ᐃᓄᓐᓄᑦ ᑐᑭᒧᐊᖅᑎᑕᐅᔪᓂ ᐃᒪᕐᓂ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑖᒃᑭᖓᓂ ᓴᕐᕙᕐᔪᐊᖅ ᐊᒻᒪ ᕿᑭᖅᑕᐃᑦ ᖃᐅᔨᓴᕐᕕᒋᔭᐅᔪᑦ.
- ᐱᓕᕆᖃᑎᒌᑦᑐᑦ ᒫᓐᓇᐅᔪᒥᑦ ᓯᕗᒧᑦᑎᐸᓪᓕᐊᔪᑦ ᑖᒃᑯᓂᖓ ᓇᔪᖅᑕᐅᔪᓂᑦ ᒥᓂᔅᑕᒧᑦ ᑎᓕᓯᔾᔪᑎᖓᓄᑦ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᖏᓐᓂᖏᑦ (MPAs) ᐊᑖᓂ *ᐃᒪᕐᓄᑦ ᐱᕐᑯᔭᖅ*, ᖁᐊᖅᑎᑦᑎᒐᔭᖅᑐᑦ ᐃᓂᖏᓐᓂ ᒫᓐᓇᐅᔪᒥᑦ ᐱᔭᕇᔭᐅᔪᓂᑦ ᑖᒃᑯᓇᓂ ᓇᖏᓐᓂᖏᓐᓄᑦ ᑎᑭᓯᒪᔪᓄᑦ ᑕᓪᓕᒪᓄᑦ ᐊᕐᕌᒍᓄᑦ.
- ᕿᑭᖅᑕᐃᑦ ᐊᒻᒪ ᓴᕐᕙᕐᔪᐊᖅ ᑲᑎᒪᔨᕋᓛᑦ, ᐃᓚᒋᔭᐅᔪᖃᖅᑐᑦ ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ, ᑲᓇᑕᐅᑉ ᒐᕙᒪᒃᑯᖏᑦ ᐊᒻᒪ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ, ᑲᑎᒪᖃᑎᖃᓚᐅᖅᑐᑦ ᐊᕕᓂᓕᓐᓂᑦ ᓄᓇᓕᓐᓂᑦ ᕿᑭᖅᑖᓗᒻᒥ ᒪᕐᕉᓐᓂ ᓄᓇᓕᓐᓂᑦ ᐳᓚᕋᕐᓂᖅᑎᓪᓗᒋᑦ ᐊᑯᓐᓂᖏᓐᓂ ᐅᑦᑑᐱᕆ 2023 ᐊᒻᒪ ᔩᓂ 2024. ᐱᔾᔪᑎᒋᔭᖏᑦ ᑖᒃᑯᓂᖓ ᑲᑎᒪᑎᑦᑎᓂᐅᔪᓂᑦ ᑐᓂᓯᓗᑎ ᑐᑭᓯᐊᕈᑎᓂᑦ ᐊᒻᒪ ᑐᓴᕝᕙᖃᕈᒪᓪᓗᑎ ᐅᖃᐅᓯᕐᓴᓂᑦ ᑐᓯᕋᕐᖑᔪᒥ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓄᑦ ᐱᔾᔪᑎᖏᑦ ᒥᓂᔅᑕᒧᑦ ᑎᓕᓯᔾᔪᑎᖓᓄᑦ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᖏᓐᓂᖏᑦ ᐊᑐᓂ ᓇᖏᓐᓂᖏᓐᓄᑦ. ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᐊᐅᓪᓚᑎᑦᑎᓂᐊᖅᑐᑦ "ᑭᓯᑦ ᑐᓴᓚᐅᖅᑕᕗᑦ" ᐅᓂᒃᑳᑦ ᑐᓴᖅᑎᑦᑎᒍᑏᑦ ᓄᓇᓕᓐᓄᑦ ᐱᑦᑎᐊᖅᓯᒪᑦᑎᐊᕋᓗᐊᕐᒪᖔᑕ ᐱᔭᐅᔪᓂᑦ ᖃᓄᐃᓐᓂᖓᔭᐃᓐᓂ ᑖᒃᑯᓂᖓ ᑲᑎᒪᑎᑦᑎᓂᐅᔪᓂᑦ ᐊᒻᒪ ᕿᓂᓂᐊᖅᑐᑦ ᑎᑎᖅᑲᒥᑦ ᐃᑲᔪᓯᖅᑕᐅᓂᖏᓐᓄᑦ ᐆᒪᔪᕐᓂᐊᑎᒃᑯᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓂ ᐊᒻᒪ Hᒻᒪᓚᐃᑦ ᑲᑎᒪᔨᖏᓐᓂ ᐊᕙᑎᓕᒫᓂᑦ ᓄᓇᓕᓐᓂᑦ.
- ᐅᕙᓂ ᔪᓚᐃ 4, 2024, ᑎᑎᖅᑲᐃᑦ ᑐᓂᐅᖅᑲᖅᑕᐅᓚᐅᖅᑐᑦ ᓕᕆᓂᖅᑲᖅᑐᓄᑦ ᐊᒻᒪ ᐊᓯᖏᓐᓄᑦ ᑎᒍᒥᐊᖅᑎᐅᖅᑲᑕᐅᔪᓄᑦ ᑐᓯᕋᖅᑐᓂ ᑐᑭᓯᐊᕈᑎᓂᑦ ᐱᔾᔪᑎᖅᑲᖅᑐᓂ ᒫᓐᓇᐅᔪᒥᑦ ᐱᓕᕆᔾᔪᑎᖏᓐᓂ ᐃᓗᐊᓂ ᑐᓯᕋᖅᖑᔪᒥᑦ ᑭᓪᓕᖓᓂ ᕿᑭᖅᑕᐃᑦ ᐊᒻᒪ ᓴᕐᕙᕐᔪᐊᖅ . ᑐᖏᓕᖏᑦ ᑎᑎᖅᑲᐃᑦ ᑐᓂᐅᖅᑲᖅᑕᐅᓂᐊᖅᑐᑦ ᕿᓂᕐᓗᑎᑎᒍᒥᐊᖅᑎᐅᖅᑲᑕᐅᔪᑦ ᐅᖃᐅᓯᖅᓴᖏᓐᓂ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓂᑦ ᐊᑐᓂ ᓇᖏᓐᓂᖏᓐᓄᑦ.

- ᐊᑕᖏᖅᑐᑎᑦ ᐃᓄᐃᑦ ᐱᔪᓐᓇᐅᑎᖏᑦ ᐊᑖᓂ ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᒻᒥ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐊᖑᓇᓯᒍᓐᓇᓂᕐᒧᑦ ᐱᔪᓐᓇᐅᑏᑦ ᐃᑉᐱᒋᔭᐅᑦᑎᐊᓂᐊᖅᑐᑦ ᓴᖅᑭᖅᑎᑕᐅᓂᖓᓄᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒥᓂᕐᑕᒧ ᑎᓕᓯᔾᔪᑎᖓᓂ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᓇᒦᓐᓂᖏᑦ ᐊᑖᓂ *ᐃᒪᕐᓄᑦ ᐱᕐᑯᔭᖅ*. ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ ᐱᔪᓐᓇᐅᑎᖏᑦ, ᐱᓕᕆᐊᓕᖏᑦ ᐊᒻᒪ ᐊᐅᓚᓂᖏᑦ ᑲᔪᓯᑎᑕᐅᓂᐊᕐᒥᔪᑦ ᐃᑉᐱᒋᔭᐅᑦᑎᐊᕐᓗᑎ.
- ᑲᑎᒪᑎᑦᑎᓂᖃᓚᐅᖅᑎᓪᓗᒋᑦ, ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᒪᓕᑦᑕᐅᔭᕆᐊᓕᓐᓄᑦ ᐊᑐᕐᓂᖓ ᑕᒪᒃᑭᓐᓄ ᓇᔪᖅᑕᐅᔪᓄ (ᐱᑕᖃᕆᐊᖃᖅᑎᑕᐅᔪᓕᒫᓂ ᐊᑖᓂ ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᑦ ᑎᑭᑕᐅᓗᑎ), ᓱᕗᓂᐊᒍᑦ ᓴᖅᑭᖅᑎᑕᐅᓂᐊᖅᑦ ᑲᓇᑕᒥ ᐅᖃᓕᒫᒐᐃᑦ, ᐃᓚᒍᑕᖓ, I.
- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᓂᕆᐅᑦᑐᑦ ᐃᓕᓴᕆᔭᐅᓯᒪᔪᓂ ᑐᓂᓯᓂᕐᒧᑦ ᕿᑭᖅᑕᐃᑦ ᐊᒻᒪ ᓴᕿᕐᔪᐊᖅ ᒥᓂᕐᑕᒧ ᑎᓕᓯᔾᔪᑎᖓᓄ ᑐᓯᕌᖑᔪᑦ ᑐᓂᔭᐅᓂᐊᖅᑐᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᖏᖅᑕᐅᓂᖓᓄ ᕼᐆᕗᐊᕆᒥ 2025 ᑲᑎᒪᓂᖓᓄᑦ.
- ᑲᑎᒪᔨᕋᖀᑦ ᑐᓐᖓᓯᑎᑦᑎᒐᔭᖅᑐᑦ ᖃᓄᐃᑦᑐᑐᐃᓐᓇᕐᓂ ᐅᖃᐅᔾᔨᒋᐊᕈᑎᓂᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓂ ᐅᕝᕙᒍᑦ ᑲᑎᒪᑎᑦᑎᓂᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᓂᖓ ᐅᕙᓗᓐᓃᑦ ᐊᓯᖏᓐᓂ ᐃᓱᒪᒋᔭᐅᔪᓂᑦ ᑐᓴᕋᖑᔪᒥᑦ ᖃᐅᔨᓴᕝᕕᒋᔭᐅᔪᓂᑦ ᓱᕗᓂᐊᒍᑦ ᑐᓂᓯᓚᐅᖅᑎᓐᓇᑕ.

2. ᑐᕝᕗᐃᖅᑐᖅ ᐃᒪᖓ ᓴᐳᒻᒥᔭᐅᓂᖓ ᓇᒦᓐᓂᖓ

- ᔫᓐᓄᐊᕆ, 2024, ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ (QIA) ᑐᓯᕋᓚᐅᖅᑐᑦ ᒑᒃᑯ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᓄᖅᑲᑎᑦᑎᓗᑎ ᐊᒻᒪ ᐊᓯᔾᔨᕐᓗᒍ ᒫᓐᓇᐅᔪᒥᑦ ᒥᓂᕐᑕᒧᑦ ᑎᓕᓯᔾᔪᑎᖓ ᐃᒪᖓ ᓴᐳᒻᒥᔭᐅᓂᖓ ᓇᒦᓐᓂᖓ ᑐᕝᕗᐃᑦᑐᖅ.
- 2019–ᒥᑦ, ᓇᒦᓐᓂᖓ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᖅ ᑕᓪᓕᒪᓄᑦ ᐊᕐᕌᒍᓄᑦ ᒥᓂᕐᑕᒧᑦ ᑎᓕᓯᔾᔪᑎᖓᓄ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖓ ᓇᒦᓐᓂᖓ (MPA) ᐊᑖᓂ *ᐃᒪᕐᓄᑦ ᐱᕐᑯᔭᖅ*. ᐱᕙᕐᓴᖅᑲᖅᑎᓪᓗᒋᑦ ᐱᓕᕆᖃᑎᖏᑦ ᐱᔭᕇᖅᓯᓗᑎ ᓈᒻᒪᓐᓂᖓᓄ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᒻᒪ ᓴᖅᑭᑎᑦᑎᓗᑎ ᐊᑯᓂᐅᔪᒧᑦ ᓴᐳᒻᒥᑎᔾᔪᑎᓂᑦ ᐊᑐᓕᕐᑯᔭᐅᒍᓂ.
- ᑲᑎᒪᑎᑦᑎᓚᐅᖅᑎᓪᓗᒋᑦ, ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐱᓚᐅᖅᑐᑦ ᐃᑲᔪᓱᖅᑕᐅᓂᕐᒧᑦ ᐆᒪᔪᕐᓂᐊᑎᒃᑯᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓂ ᐊᒻᒪ Hᐊᒻᒪᓚᐃᑦ ᑲᑎᒪᔨᖏᓐᓂ ᑕᓪᓕᒪᓕᒫᓂ ᐊᑦᑐᐊᓂᓕᓐᓄᑦ ᖁᑦᑎᑦᑐᒥ ᐅᑭᐅᑕᖅᑐᒥ ᓄᓇᓕᖕᓂᑦ (ᐃᑉᐱᐊᕐᔪᒃ, ᑲᖏᖅᑐᒑᐱᒃ, ᐊᐅᔪᐃᑦᑐᖅ, ᒥᑦᑎᒪᑕᓕᒃ, ᖃᐅᔨᐃᑦᑐᖅ)ᐊᐅᔭᖓᓂ 2024.
- ᒫᑦᓯᒥ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐊᒻᒪ ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑕᑯᓵᐅᑎᑦᑎᖃᑎᒌᓚᐅᖅᑐᑦ ᑐᖏᓕᖓᓄ ᒥᓂᕐᑕᒧᑦ ᑎᓕᓯᔾᔪᑎᖓᓄ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᓇᒦᓐᓂᖓ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ ᐊᖏᖅᑕᐅᓗᓂ, ᐱᔭᐅᓚᐅᖅᑐᖅ ᐃᐳᕈᒥ.
- ᐅᕝᕙᓂ ᐋᒡᒋᓯ 14, 2024, ᓄᑖᖅ ᒥᓂᕐᑕᒧᑦ ᑎᓕᓯᔾᔪᑎᖓᓄ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᓇᒦᓐᓂᖓ (ᑎᓕᓯᔾᔪᖓ ᓈᓴᐅᑎᖓ 2 ᓇᓗᓇᐃᖅᑕᐅᓂᖓ ᑐᕝᕗᐃᑦᑐᖅ ᐃᒪᖓ ᓴᐳᒻᒥᔭᐅᓂᖓ ᓇᒦᓐᓂᖓ) ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᑲᓇᒥ ᐅᖃᓕᒫᒐᕐᓂᑦ, ᐃᓚᒍᑕᖓ II ᐱᔭᕇᖅᓯᔪᖅ ᓇᓗᓇᐃᖅᑕᐅᓂᖓᓄᑦ ᐱᕝᕕᓕᐊᓂᖓ ᒫᓐᓇᐅᔪᒧᑦ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᓇᒦᓐᓂᖓ.
- ᑖᒃᑯᐊ ᓄᑖᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᓖᑦ ᐱᑕᖃᖅᑎᑦᑎᔪᑦ ᑕᓪᓕᒪᒃᑲᓂᕐᓂᑦ ᐊᕐᕌᒍᓂᑦ ᓴᐳᒻᒥᔭᐅᓂᖓᓄᑦ ᑐᕝᕗᐃᑦᑐᖅ ᐱᓕᕆᖃᑎᖏᑦ ᐃᖅᑲᓇᐃᔭᖃᑎᖃᖅᑎᓪᓗᒋᑦ ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓂ ᐃᓱᒪᒋᓗᒍ ᐃᓄᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᐊᒻᒪ ᓴᐳᒻᒥᔭᐅᔪᖅ ᓇᒦᓐᓂᖓ ᓇᒦᓐᓂᖓᓄᑦ ᐊᑯᓂᐅᔪᒧᑦ.

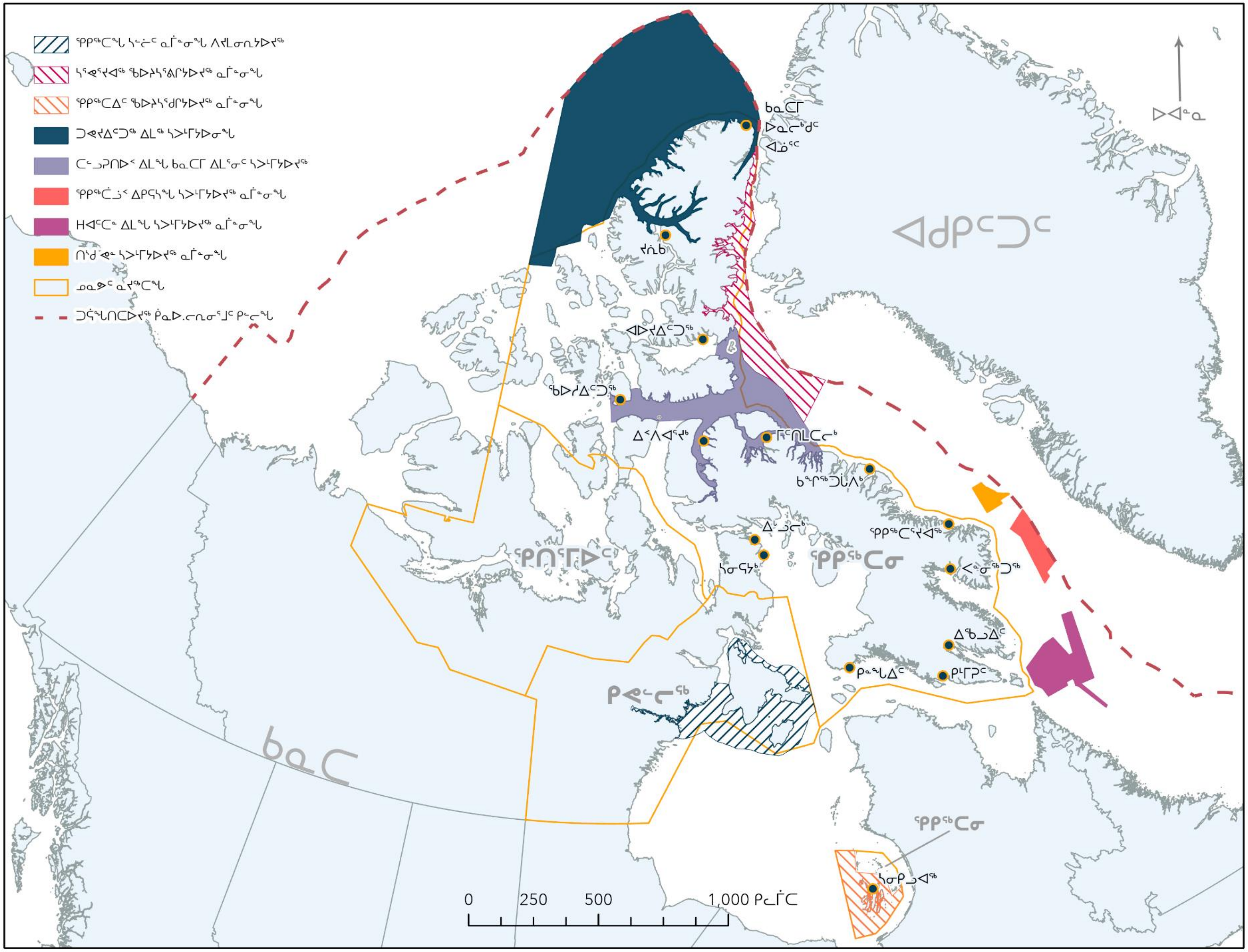
3. ᓴᓪᓖᑦ ᕿᑭᖅᑕᖓ ᓇᒦᓐᓂᖓ ᐱᔪᒪᓂᕆᔭᐅᔪᖅ

- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐃᖅᑲᓇᐃᔭᖃᑎᖃᖅᑐᑦ ᕿᑭᖅᑖᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓂ ᐊᒻᒪ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖓᓂ ᓱᕗᒧᑦᑎᓪᓗᒍ ᓴᓪᓖᑦ ᕿᑭᖅᑕᖓ ᓇᒦᓐᓂᖓ ᐱᔪᒪᓂᕆᔭᐅᔪᖅ ᓇᓗᓇᐃᖅᑕᐅᔪᓐᓇᕐᓂᖓᓄ ᓄᑖᖑᓪᓗᓂ ᐃᓴᕐᓄ ᐱᕐᑯᔭᖅ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᓇᒦᓐᓂᖓ.
- ᓴᓪᓖᑦ ᕿᑭᖅᑕᖓ ᓇᒦᓐᓂᖓ ᐱᔪᒪᓂᕆᔭᐅᔪᖅ ᐱᖃᓯᐅᔾᔨᓂᖃᖅᑐᖅ ᓄᓇᖅᐸᓯᒻᒥ ᐃᒪᖓ ᖃᓂᒋᔭᖏᓐᓂ ᓴᓪᓖᑦ ᕿᑭᖅᑕᖓ ᐊᒻᒪ ᐃᓪᓗᓕᒫᕐᔪᒃ ᕿᑭᖅᑕᓂ ᓄᓇᕗᑦ. ᑭᖑᓪᓕᖅᐹᒥ ᑭᓪᓕᖓ ᐱᑕᖃᕈᓐᓇᖅᑐᓂᑦ ᓱᕗᓂᒃᓴᒥ ᐃᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᒦᓐᓂᖓ ᑐᓐᖓᓂᖃᓂᐊᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᐊᒻᒪ ᑲᑎᒪᑎᑦᑎᓂᐅᔪᓂᑦ.
- ᐊᑕᖏᖅᑐᑎ ᐃᓄᐃᑦ ᐱᔪᓐᓇᐅᑎᖏᑦ ᐊᑖᓂ ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐊᖑᓇᓯᒍᓐᓇᓂᕐᒧᑦ ᐱᔪᓐᓇᐅᑏᑦ ᐃᑉᐱᒋᔭᐅᑦᑎᐊᓂᐊᖅᑐᑦ ᓴᖅᑭᖅᑎᑕᐅᓂᖓᓄᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒥᓂᕐᑕᒧ ᑎᓕᓯᔾᔪᑎᖓᓂ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᑦ ᓇᒦᓐᓂᖏᑦ ᐊᑖᓂ *ᐃᒪᕐᓄᑦ ᐱᕐᑯᔭᖅ*. ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ ᐱᔪᓐᓇᐅᑎᖏᑦ, ᐱᓕᕆᐊᓕᖏᑦ ᐊᒻᒪ ᐊᐅᓚᓂᖏᑦ ᑲᔪᓯᑎᑕᐅᓂᐊᕐᒥᔪᑦ ᐃᑉᐱᒋᔭᐅᑦᑎᐊᕐᓗᑎ.
- ᓇᓗᓇᐃᔭᖅᑕᐅᖅᑲᐅᔪᑦ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ ᑭᖑᓪᓕᖅᐹᒥ ᑲᑎᒪᓂᖅᑲᖅᑎᓪᓗᒋᑦ, ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ, ᑭᕙᓪᓕᖅ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒋᖏᑦ ᐊᒻᒪ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐃᓚᒋᔭᐅᑎᑦᑎᓚᐅᖅᑐᑦ ᑕᓪᓕᒪᓂᑦ ᓄᓇᓕᖕᓂᑦ ᖃᓂᒋᔭᖓᓂ ᓴᓪᓖᑦ ᕿᑭᖅᑕᖓᓂ ᓇᒦᓐᓂᖓ ᐱᔪᓂᕆᔭᐅᔪᖅ (ᖃᒪᓂᑦᑐᐊᖅ, ᐃᓪᓗᓕᒫᕐᔪᒃ, ᓴᓪᓖᑦ, ᓇᐅᔾᔮᑦ ᐊᒻᒪ ᑲᖏᕐᖠᓂᖅ) ᐅᑭᐅᖓᓂ 2023-2024, ᑐᓂᓯᓪᓗᑎ ᐊᒻᒪ ᕿᓂᖅᑐᑎᑦ ᐅᖃᐅᓯᖕᓴᓂᑦ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᒦᓐᓂᖓᓄ ᑐᓯᕋᐅᑦ.
- ᑎᒍᒥᐊᖅᑎᐅᖃᑕᐅᔪᑦ ᐃᓚᒋᔭᐅᑎᑕᐅᓚᐅᕐᒥᔪᑦ ᑎᑎᕋᖅᑕᐅᕆᐊᓚᖅᓯᒪᔪᓂ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖏᓐᓄᑦ ᓇᒦᓐᓂᖓ ᑐᓯᕋᐅᒪᒥᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐱᓕᕆᖃᑦᑕᖅᑐᑦ ᐅᔭᕋᓐᓂᐊᓂᕐᒥᑦ ᐊᒻᒪ ᐅᖅᓱᐊᓗᓐᓂᑦ, ᐅᒥᐊᕐᔪᐊᓕᕆᓂᕐᒥᑦ, ᐊᒻᒪ ᐳᓚᕋᖅᑐᓕᕆᓂᕐᒥᑦ.

- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᑐᓴᓚᐅᖅᑐᑦ ᐅᕐᑲᐅᓯᔅᓴᓂᑦ ᓇᒦᓐᓂᖅᒐᓄᑦ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᑐᕌᒐᕐᕆᓐᓂ; ᑐᔅᓯᕚᖑᓯᒪᔪᖅ ᑭᓪᓕᖅᑲ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖅᑭᓐᓄᑦ ᓇᒦᓐᓂᖅᑲ, ᐊᒻᒪ ᑐᔅᓯᕚᖑᓯᒪᔪᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᑦ ᒪᓕᑦᑕᐅᔭᕆᐊᕐᑲᖅᑎᑦᑎᒍᓐᓇᖅᑐᑦ, ᑭᓪᓕᕐᑲᖅᑎᑦᑎᓗᑎ, ᐱᔪᓐᓇᖅᑎᑕᐅᓗᑎ ᖃᓄᐃᓐᓂᕐᑲᕆᐊᕐᑲᕐᓗᑎ, ᐅᕝᕙᓘᓐᓃᑦ ᐱᔪᓐᓇᖅᑎᑕᐅᓗᑎ ᐱᓕᕆᔭᐅᔪᑦ, ᐃᑉᐱᒍᓱᑦᑎᐊᕐᓂᕐᑲᖅᑐᑎᑦ ᓄᓇᕗᒻᒧᑦ ᐊᖏᕈᒻᒥᑦ. ᑐᓴᖅᑎᑦᑎᒍᑎᐅᔪᑦ ᒫᓐᓇᐅᔪᒥᑦ ᑲᑎᖅᓱᖅᑕᐅᔪᑦ ᐅᕗᖓ “ᑭᓯᐃᑦ ᑐᓴᓚᐅᖅᑕᕗ” ᐅᓂᒃᑳᓂ ᐊᒻᒪ ᐊᐱᖅᑯᑎᓄᑦ ᐊᒻᒪ ᑭᐅᔾᔪᑎᓄᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ ᓴᖅᑭᖅᑎᑕᐅᕙᓪᓕᐊᔪᑦ ᑐᓂᔭᐅᓂᐊᖅᑐᑦ ᑕᐃᒃᑯᓄᖅᑲ ᐃᓚᒋᔭᐅᕐᑲᑕᐅᓚᐅᖅᑐᓄᑦ ᑲᑎᒪᑎᑦᑎᓂᐅᔪᓄᑦ.
- ᑖᒃᑯᐊ ᑐᓴᖅᑎᑦᑎᒍᑏᑦ ᕐᑲᐅᔨᔾᔪᑎᐅᓂᐊᖅᑐᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖅᑲᓄᑦ ᑮᒍᓪᓕᖅᐹᒥ ᐃᒪᐃᑦ ᓴᐳᒻᒥᔭᐅᓂᖅᑭᓐᓄᑦ ᓇᒦᓐᓂᖅᑲ ᑐᔅᓯᕋᐅᒻᒧᑦ ᐊᒻᒪ ᓄᓇᓕᓐᓅᖅᑕᐅᓗᑎ ᑐᖅᑭᓕᖅᑲᓂ ᓄᓇᓕᓐᓄᑦ ᐳᓚᕋᕐᑲᑦᑕᓂᕐᒧᑦ, ᐸᕐᓇᑕᐅᓯᒪᔪᖅ ᐅᑭᐅᖅᑲᓂ 2024-2025 ᑭᕙᓪᓕᕐᒥ ᓄᓇᓕᓕᒫᓄᑦ.
- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᐊᒻᒪ ᑭᕙᓪᓕᕐᒥ ᐃᓄᐃᑦ ᑲᑐᔾᔨᕐᑲᑎᒌᑦ ᐃᖅᑲᓇᐃᔭᕐᓂᕐᑲᖅᑐᑦ ᐋᖅᑭᒋᐊᖅᓯᒪᔪᓂᑦ ᐱᒻᒪᕆᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᓚᐅᖅᓯᒪᖅᑭᑦᑐᓂᑦ.

ᑎᑎᕋᖅᑐᖅ: ᐃᒪᕐᒥ ᐸᕐᓇᐃᓂᕐᒧᑦ ᐊᒻᒪ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᐊᒻᒪ ᐃᕐᑲᓗᓕᕆᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ, ᐅᑭᐅᑕᖅᑐᒥ -- ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ.

ᐅᓪᓗᖅᑲ: ᐋᒡᒌᓯ 8, 2024



ᐃᓚᒌᑦᑕᖅ 1. ᐱᑕᖃᕈᓐᓂᖅᑐᑦ ᐊᒻᒪ ᐱᑕᖃᕈᓐᓇᖅᑐᑦ ᓯᕗᓂᐊᓂ ᐅᑭᐅᑕᖅᑐᒥ ᓴᐳᒻᒥᔭᐅᔪᑦ ᓇᒦᓐᓂᖏᑦ.

ᐊᖏᕈᑎᒃᓴᐃᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄ – ᐊᑯᓕᕈᕐᕕᒃ 8, 2024

ᐅᑯᓄᖓ: ᑐᓴᒃᓴᖅ: *X* **ᐃᓱᒪᓕᐊᕆᔭᐅᔪᑦ:**

ᐅᖃᐅᑕᐅᔪᖅ: ᒫᓐᓇᒨᓕᖓᔪᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᖅ ᐅᓇ ᐱᕙᑎᒋᓪᓗᒍ **ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓᓂ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᓂᒃ ᓴᐳᔾᔨᕕᐅᔪᖅ**

ᐱᕙᑎ

ᑐᑭᓯᒋᐊᕐᑎᑦᑎᓂᖅ ᒫᓐᓇᒨᓕᖓᔪᓂᒃ ᐆᒥᖓ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᖅ (IMP) ᐅᓇ ᐱᕙᑎᒋᓪᓗᒍ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓᓂ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᓂᒃ ᓴᐳᔾᔨᕕᐅᔪᖅ (TINMCA) ᐃᓚᒋᓪᓗᒍ ᒫᓇ ᐱᓕᕆᐊᖑᔪᓂᒃ, ᓄᓇᓕᖕᓄ ᑐᓴᕆᐊᕐᓂᐅᓵᓚᐅᕐᑐᒥᒃ, ᖃᓄᐃᓕᐅᕆᐊᖃᕐᓂᕆᓴᕐᑎᓐᓂᒃ ᐊᒻᒪ ᖃᑯᒍ ᐱᔭᕇᓛᕐᓂᖓᓂᒃ. ᑭᖑᓪᓕᕐᒥᑦ ᑐᓴᕐᑎᑦᑎᓂᐅᓚᐅᕐᑐᖅ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄ ᐅᐱᕐᖔ 2020–ᐅᑎᓪᓗᒍ. ᑭᖑᕚᕐᓂᖃᓚᐅᕐᑐᒍ ᐱᔭᕇᕐᓱᒃᓗᑎᒍ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᓂᒃ, ᑭᓯᐊᓂ ᐸᕐᓇᐃᔨᑦ ᑲᑎᒪᔨᖏᑦ ᐱᓕᕆᑦᑎᐊᓚᐅᕐᑐᑦ ᓯᕗᒧᐊᒍᑎᓂᒃ ᐊᕐᕌᒍᒧ ᐊᒻᒪ ᓂᕆᐅᖕᓂᖃᕐᓗᑎᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᓂᒃ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᓗᑎᒃ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᓂᒍᕐᑎᑕᐅᓗᑎᒃ ᑕᖅᑭᕐᓂᒃ ᐊᕗᓐᓂᕕᒃ ᐊᒻᒪ ᒪᓐᓃᑦ 2025 –ᖑᓕᕐᐸᑦ ᑲᑎᒪᓂᐅᓂᐊᕐᑐᒥᒃ.

ᐱᓕᕆᐊᖑᓚᐅᕐᑑᑉ ᑐᑭᖏᑦ

ᐱᓕᕆᕙᒡᔪᐊᓚᐅᕐᑐᑦ ᓴᕿᑎᑦᑎᓪᓗᑎᒃ ᐊᑐᕐᑕᐅᒋᐊᓕᖕᓂᒃ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓᓂ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᓂᒃ ᓴᐳᔾᔨᓯᒪᐊᕐᓂᖏᓐᓂᒃ ᒪᓕᒃᓗᒋᑦ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᔾᔭᐅᓯᒪᓂᖏᑕ ᐃᓂᖓᓂ ᐱᖁᔭᖅ (2002) ᒪᓕᒃᓗᒋᑦ ᐅᑯᐊᖑᔪ: ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐱᓕᕆᐊᒃᓴ ᓈᒻᒪᒃᑲᓗᐊᕐᒪᖔᑦ ᐊᒻᒪ ᐋᑎᖃᕐᓂᐊᕋᓗᐊᕐᒪᖔ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᓂᒃ ᓴᐳᔾᔨᓯᒪᐊᕐᓂᖏᓐᓂᒃ (ᐱᔭᕇᑕᐅᓚᐅᕐᑐᒥᒃ 2017-ᖑᑎᓪᓗᒍ); ᐋᔩᖃᑎᒌᖕᓂᖅ ᐆᒥᖓ ᐃᓄᐃᑦ ᐊᒃᑐᖅᑕᐅᓂᐊᕐᓂᖏᓐᓄᑦ ᐃᑲᔫᓯᐊᖅᑖᕈᓐᓇᐅᑎᓄᑦ ᐊᖏᕈᑎᓂᒃ (ᐱᔭᕇᓚᐅᕐᑐᒥᒃ 2019-ᖑᑎᓪᓗᒍ); ᓴᕿᑎᓂᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᓂᒃ (ᐱᓕᕆᐊᕆᔭᐅᓕᕐᑐᖅ); ᐊᒻᒪᓗ ᓴᕿᑎᓗᑎᒃ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᓂᒃ ᓴᐳᔾᔨᕕᒃ ᒪᓕᒐᖑᕐᑎᑕᐅᓗᑎ (ᐱᓕᕆᐊᕆᔭᐅᓕᕐᑐᖅ).

ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ 108,000 km^2 ᐊᖏᓂᖃᖅᑐᑦ ᕿᑭᖅᑕᓂ ᐅᐊᖕᓇᖓᓂ ᐃᓂᖃᖅᑐ, ᐊᒻᒪᓗ ᓄᓇᐅᑉ ᐃᓚᖓᓂᑦ (ᐱᖓᓱᓂᒃ ᑎᖕᒥᐊᓂᒃ ᐃᓐᓅᕈᕐᒥᐅᑕᖃᐅᖅᑐᖅ) ᐊᒻᒪ ᕿᑭᖅᑕᓕᒫᖏᑦ ᑐᖕᒫᓂ 400–ᒦᑕᓂᒃ ᐳᖅᑐᓂᖃᖅᑐᑦ). ᓄᓇᓖᑦ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐃᓚᓕᐅᑎᓯᒪᔪᑦ ᐅᑯᐊ: ᐊᐅᔪᐃᑦᑐᖅ, ᖃᐅᓱᐃᑦᑐᖅ, ᐃᒃᐱᐊᕐᔪᒃ, ᒥᑦᑎᒪᑕᓕᒃ ᐊᒻᒪ ᑲᖏᖅᖢᒑᐱᒃ.

ᐸᕐᓇᐃᔨᑦ ᑲᑎᒪᔨᖏᑦ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓᓂᒃ, ᒪᕐᕉᖕᓂᒃ ᑭᒡᒐᖅᑐᐃᔩᖃᖅᑐᑦ ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᑯᓐᓂᖕᒫᖅᑐᓂᒃ, ᐊᑕᐅᓯᕐᒥᒃ ᑭᒡᒐᖅᑐᐃᔩᖃᖅᓗᑎᒃ ᑲᓇᑕᒥ ᒥᕐᖑᐃᕐᓯᕐᕕᓕᕆᔨᒃᑯᓐᓂ; ᐊᒻᒪ ᐊᑕᐅᓯᕐᒥᒃ ᑭᒡᒐᖅᑐᐃᔩᖃᖅᓗᑎᒃ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓂᖔᖅᑐᒥᒃ, ᐱᓕᕆᐊᒃᓴᖃᖅᓗᑎᒃ ᓯᕗᓕᖅᑎᐅᓪᓗᑎᒃ ᓴᕿᑎᒋᐊᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᓕᕆᓪᓗᑎᒃ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᔾᔭᐅᓯᒪᓂᖏᑕ ᐃᓂᖓᓂᒃ. ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᖅ ᐊᑐᕐᑕᓂᐊᕐᓗᓂ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᔾᔭᐅᒐᓱᐊᕐᑕᐅᔪᒥᒃ ᑕᓪᓕᒪᓄᑦ ᐊᕐᕌᒍᓄᑦ ᐸᕐᓇᐃᑎᓪᓗᒋᑦ ᐊᕐᕌᒍ 10–ᓂᒃ ᐃᓗᐃᑦᑐᒥ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓴᕿᑎᒐᓱᐊᕐᓂᖏᓐᓂᒃ. ᓴᕿᑎᓂᐅᔪᖅ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᓂᒃ ᐅᓄᕐᑐᓂᒃ ᑐᓴᕐᑎᑕᐅᓚᐅᕐᓂᑦᑎᓐᓄᑦ ᓄᓇᓕᖕᒥᐅᓂᒃ, ᐊᒻᒪ ᑎᒍᒥᐊᕐᑎᐅᔪᓂᒃ ᑐᓴᕐᑎᑕᐅᓪᓗᑕ ᐃᓄᖕᓂᓪᓗ.

ᐸᕐᓇᒃᑕᐅᓯᒪᔪᖅ ᐃᓗᓕᖃᖅᑐᖅ **ᑕᑯᓐᓇᕐᓗᒍ** ᓇᓗᓇᐃᕆᓗᑎᒃ ᓯᕗᓂᒃᓴᒥ ᖃᓄᐃᓕᖓᓂᐊᕐᓂᖏᓐᓄᑦ ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᔾᔭᐅᒋᐊᖃᕐᓂᖏᑕ ᐃᓂᖏᓂᒃ ᐊᕐᕌᒍᓄ 15-20–ᓄ; ᐊᐅᓚᑦᑎᓂᕐᒧ **ᑐᕌᒐᐃᑦ** ᓇᓗᓇᐃᕆᓯᒪᔪᑦ ᐱᓕᕆᐊᒃᓴᐃᑦ ᐱᔭᕇᖅᑎᐊᖅᑯᓪᓗᒋᑦ ᐸᕐᓇᒃᑕᐅᓯᒪᔪᑦ ᒪᓕᒃᓗᒋᑦ, ᐃᓚᖃᖅᓗᑎᒃ **ᑎᑭᑕᐅᔪᒪᔪᓂᒃ** ᑐᑭᓕᐅᑦᑎᐊᕐᓯᒪᔪᓂᒃ ᖃᐅᔨᓴᕐᓂᐊᕐᓗᑎᒍ ᑎᑭᒐᓱᐊᕐᑕᑦᑕ

ᑐᕌᑦᑎᐊᓕᕋᓗᐊᕐᒪᖔᑕ; ᐊᒻᒪ **ᑭᒡᓕᓕᐅᕈᑎᓂᒃ** ᐸᕐᓇᐃᓂᖅ ᓇᓗᓇᐃᕆᓂᐊᕐᒪᑦ ᐊᐅᓚᑦᑎᔾᔪᑏᑦ ᑐᕌᒐᖕᒋᓐᓂᒃ ᐊᑦᔩᖕᒋᑦᑐᓂᒃ ᐃᓂᖃᕐᕕᐅᔪᓂᒃ ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᑕᐃᑦ ᓴᐳᑦᔭᐅᒋᐊᖃᕐᓂᖕᒋᑦ ᐊᕕᒃᑐᕐᓗᒋᑦ ᓇᓗᓇᐃᕆᑦᑎᐊᕐᓗᒋᓪᓗ ᑭᒡᓕᒋᓂᐊᕐᑕᖕᒋᓐᓂᒃ. ᑭᒡᓕᓕᐅᕆᓃᑦ ᑐᑭᓕ ᖃᓄᐃᑦᑐᓂᒃ ᐱᓕᕆᐊᒃᓴᓂ ᓴᕿᑦᑎᓂᐊᕐᒪᖔᑦᑕ ᐃᓂᒋᔭᐅᔪᓂᒃ, ᐊᒻᒪ ᖃᓄᖅ ᐊᑐᓕᖁᔨᓂᕐᓂᒃ ᐃᓯᒪᒃᓴᕐᓯᐅᕆᐊᖃᕐᓂᐅᔪᑦ. ᐃᓄᐃ ᐱᔪᓐᓇᐅᑏᑦ–ᑐᓐᖓᕕᒋᓗᒋᑦ ᐊᖑᓇᓯᒡᕕᐅᔪᓐᓇᕐᑐᖅ ᐊᒻᒪ ᐱᖁᓯᕆᔭᐅᔪᑎᒍ ᐊᑐᕐᑕᐅᔪᓐᓇᕐᓗᑎᒃ ᑭᒡᓕᓕᐅᕐᑕᐅᓯᒪᒪᒐᓗᐊᕈᑎᒃ, ᑕᐃᒪᓐᖓᓕᒫᖅ. ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᔾᔭᐅᓯᒪᔪᑦ ᐃᓂᖕᒋᑦ ᐃᓗᓕᖃᕐᑐᖅ ᐅᑯᓂᖓ:

- ᑭᒡᓕᓕᐅᕐᓯᒪᔪᑦ ᓴᐳᑎᔭᐅᓯᒪᕙᒃᔪᐊᕐᑐᑦ ᓴᐳᑎᒋᐊᕐᓗᒋᑦ ᐃᕐᓯᓴᕋᐃᑦᑐᓂᒃ ᐸᕝᕕᒍᓯᒃᓴᕋᐃᑦᑐᓂᒃᓗ (ᐆᒃᑑᑎᒋᓗᒋᑦ ᐃᓐᓈᕈᕐᒦᑦᑐᓂᒃ ᑎᖕᒥᐊᑦ). ᑭᓯᐊᓂ ᐃᓄᐃᑦ ᐅᐸᒍᓐᓇᕐᑕᖕᒋᑦ ᐊᒻᒪ ᐊᒥᓲᖕᒋᑦᑐᑦ ᖃᐅᔨᓴᕐᑎᓄᑦ ᐅᐸᒃᑕᐅᔪᓐᓇᕆᓪᓗᑎᒃ ᐃᓛᓐᓂᒃᑯᑦ.
- ᑭᒡᓕᓕᐅᕐᓯᒪᔪᑦ ᓴᐳᔾᔨᓪᓗᑎᒃ ᐆᒪᔪᐃᑦ ᐃᓂᒋᕙᒃᑕᖕᒋᓐᓂᒃ (ᐆᒃᑑᑎᒋᓗᒋᑦ, ᐊᖁᓵᕐᕕᐅᖃᑦᑕᕐᑐᑦ ᓂᕐᔪᑎᓄᑦ, ᒥᒃᑭᑦᑐᒑᒥᓂᒃ ᐱᒪᑐᒃᑐᑦ) ᐱᓕᕆᐊᒃᓴᐃᑦ ᐃᓚᖕᒋᓪᓗ ᐅᐸᒃᑕᐅᔭᕆᐊᖃᓐᖕᒋᒻᒪᕆᒃᑐ ᐱᐅᓐᖕᒋᑦᑐᒥᒃ ᑐᓗᕐᑕᕈᑎᖃᕋᓴᕐᓂᖕᒋᓐᓄᑦ ᐆᒪᔪᓂᒃ;
- ᐊᒥᓱᓂᒃ ᐃᓂᒋᔭᐅᔪᒥᒃ ᐊᑦᔩᓐᖕᒋᑦᑐᓂᒃ ᐱᓕᕆᐊᒃᓴᓄ ᐊᑐᕐᑕᐅᕙᒃᑐᓂᒃ ᐊᒻᒪ ᐊᑐᕐᑕᐅᔪᓐᓇᕐᑐᑦ ᐱᔪᓐᓇᐅᑎᖃᕐᑎᑕᐅᓂᐊᕐᑐᑦ ᐱᐅᓐᖕᒋᑦᑐᒥᒃ ᑐᓗᕐᑕᕈᑎᓕᐅᖕᒋᒃᑯᑎ ᐃᒪᕐᒥᐅᑕᐃ ᐊᕙᑎᖕᒐᓂᒃ ᐅᕝᕙᓘᓐᓃᑦ ᐃᓄᐃᑦ ᐱᔪᓐᓇᐅᑎᖃᕐᑎᓄᑦ, ᐊᒻᒪᓗ
- ᐃᓂᒋᔭᐅᔪᑦ ᓇᓗᓇᐃᕐᑕᐅᓯᒪᔪᑦ ᐱᒻᒪᕆᒋᔭᐅᓂᖕᒋᑦ ᓄᓇᓕᖕᒥᐅᓄᑦ, ᐅᔾᔨᕐᑐᕐᑕᐅᓂᐊᕐᑐᑦ ᐊᐅᓚᑕᐅᓗᑎᒃᓗ ᐃᓄᐃᑦ ᐊᑐᖃᑦᑕᕐᑕᖕᒋᓐᓂᒃ ᑐᓗᕐᑕᕈᑎᖃᕐᓂᐊᖕᒋᒻᒪᑕ.

ᑐᓴᕆᐊᕐᓂᐅᔪᑦ

- ᑐᓴᕆᐊᕐᓯᒪᓂᐅᓚᐅᕐᑐᑦ ᐊᐅᓚᑦᑎᓂᕐᒧ ᐸᕐᓇᐅᑎᐅᓚᐅᑲᒃᑐᓂᒃ ᕿᒥᕐᕈᓚᐅᕐᑐᑦ ᐊᕐᕌᒍ 2018 ᐊᒻᒪ 2019-ᖑᑎᓪᓗᒍ ᐅᑯᓂᖓ ᓄᓇᓕᖕᒥᐅᓂ ᐊᐅᓱᐃᑦᑐᖅ, ᖃᐅᓱᐃᑦᑐᖅ, ᐃᒃᐱᐊᕐᔪᒃ, ᒥᑦᑎᒪᑕᓕᒃ ᐊᒻᒪ ᑲᖕᒋᕐᑐᒑᐱᖕᒥᑦ ᑐᓴᕐᕕᐅᔭᕐᑐᖅᓗᑎᒃ ᑭᓯ ᐸᕐᓇᐅᑎᐅᓚᐅᑲᒃᑐᓂᒃ ᐃᓚᓯᔪᒪᒻᒪᖔᑕ.. ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔩᖕᒋᑦ ᑲᑎᒪᖃᑎᖃᓚᐅᕐᑐᑦ ᕼᐊᒻᒪᓚᒃᑯᓐᓂ, ᐆᒪᔪᓕᕆᔨᐊᒑᓂᒃ (HTO), ᐊᒻᒪ ᐊᓯᖕᒋᓐᓂᒃ ᓄᓇᓕᖕᒥᐅ ᑎᒥᖁᑎᖕᒋᓐᓂᒃ ᐊᒻᒪᓗ ᑭᑐᒥᐊᓄᑦ ᑐᑭᓯᑎᑦᑎᓂᕐᒥᒃ ᑲᑎᒪᑎᑦᑎᓪᓗᑎ. ᐃᒻᒪᖄ 300–ᑭᑐᒥᐊᑦ ᑐᓴᕐᕕᐅᓚᐅᕐᑐᑦ. ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔩᖕᒋᑦ ᑐᓴᕐᑕᒥᓂᒃ ᑲᑎᒪᖃᑎᖃᕐᑎᓪᓗᒋᑦ ᐃᓚᓕᐅᕆᓚᐅᕐᑐᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᑲᒃᑐᒧᑦ.

- ᓇᑦᑎᐊᑦ 2024-ᖑᑎᓪᓗᒍ, ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔩᖕᒋᑦ ᐊᐅᓪᓚᓚᐅᕐᑐᑦ ᓄᓇᓕᖕᓄ ᐅᑯᓄᖓ: ᐊᐅᔪᐃᑦᑐᖅ, ᖃᐅᓱᐃᑦᑐᖅ, ᐃᒃᐱᐊᕐᔪᒃ, ᒥᑦᑎᒪᑕᓕᒃ, ᐊᒻᒪ ᑲᖕᒋᖅᑐᒑᐱᒃ, ᐅᓂᒃᑳᕆᐊᕐᓗᑎᒃ ᑐᓴᓚᐅᕐᑕᒥᓂᒃ ᓇᓕᖅᑯᑦᑎᒐᓗᐊᕐᒪᖔᑕ ᐊᑕᖕᒋᕐᓗᒋᑦ ᓄᓇᓕᖕᒥᐅᑦ ᐃᓱᒫᓘᑎᖕᒋᓐᓂᒃ ᐊᒻᒪ ᓯᕗᓪᓕᐅᑎᒋᐊᓕᖕᓂᒃ ᐃᓕᔭᐅᑦᑎᐊᕋᓗᐊᕐᒪᖔᑖ ᐸᕐᓇᐅᑎᐅᓚᐅᑲᒃᑐᒥᑦ ᐸᕐᓇᐃᑎᓪᓗᒋᑦ. ᑲᑎᒪᖃᑎᖃᓚᐅᕐᑐ ᐅᑯᓂᖓ: ᕼᐊᒻᒪᓚᑦᑯᑦ ᑲᑎᒪᔩᖕᒋᓐᓂᒃ, ᐆᒪᔪᓕᕆᔨᐊᒑᓂᒃ, ᓄᓇᓕᖕᓂᑦ ᓄᓇᓕᕆᓂᕐᒧᑦ ᑲᑎᒪᔩᕋᒑᖕᒋᓐᓂᒃ ᐊᒻᒪ ᓇᐅᑦᑎᖅᓱᖅᑎᓂᒃ ᐅᓪᓗᕐᓄ ᒪᕐᕉᖕᓂᑦ. ᑲᑎᕝᕕᖃᕐᑎᑦᑎᓚᐅᕆᓪᓗᑎᒃ ᓄᓇᓕᓕᒫᓂᒃ. ᐃᒻᒪᖄ 130–ᑭᑐᒥᐊᑦ ᐅᐸᒎᑎᓚᐅᕐᑐᑦ ᑲᑎᒪᓂᐅᔪᒧᑦ. ᑐᓴᕐᕕᐅᔭᕐᑐᖅᑎᓪᓗᒋᑦ, ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔩᖕᒋᑦ ᑐᓴᓚᐅᕐᑐᑦ ᑭᓯ ᓴᐳᔾᔭᐅᒋᐊᖃᕐᓂᖕᒋᓐᓂᒃ ᐊᒻᒪᓗ ᖁᐊᕐᓵᕐᓇᕐᑐᖃᕐᓂᐊᓂᒃ ᑕᐃᑲᓂ ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᑦᔭᐅᓯᒪᔫᑉ ᐃᓂᖓᓂ (ᐆᒃᑑᑎᒋᓗᒋᑦ: ᐃᓱᒫᓘᑕᐅᓚᐅᕐᑐᑦ ᐅᒥᐊᕐᔪᐊᑦ ᑎᑭᐸᒃᑐᑦ ᓇᐅᒃᑯᑦ ᐃᖕᒋᕐᕌᓂᖕᒋᑦ ᐃᓕᖁᓯᐊᓂᒃ, ᐊᒻᒪ ᑐᓗᕐᑕᕈᑎᓕᐅᕐᓂᖕᒋᓐᓂᒃ, ᐅᓄᓗᐊᕐᑐᓂᒃ ᑭᑐᒥᐊᓄᑦ ᐊᑦᑕᕐᓇᕈᑕᐅᔪᓐᓇᕐᓂᖕᒋᓐᓂᒃ ᐅᕝᕙᓘᓐᓃᑦ ᐊᕙᑎᒧ ᐱᕿᓗᐊᕿᓂᕐᓂᒃ). ᑲᑎᒪᔩᑦ ᑐᓴᓚᐅᕐᑕᒥᓂᒃ ᐊᖀᒋᐊᓕᖅᓗᑎᒃ ᑐᕌᒐᓕᔭᐅᔪᓂᒃ ᐊᒻᒪ ᑎᑭᑕᐅᔪᒪᔪᓂᒃ ᐃᓚᒋᓪᓗᒍ ᑭᒡᓕᓕᐅᕈᑎᑉ ᐸᕐᓇᐅᑎᖕᒋᓐᓂᒃ. ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔩᖕᒋᑦ ᐅᖃᐅᑎᔭᐅᓚᐅᕆᓇᑦ ᐋᕿᒋᐊᕐᑕᐅᓗᑎᒃᓗ ᑕᑯᓐᓇᕐᑕᐅᔫᑉ ᐅᖃᐅᓯᖕᒋ ᐊᒻᒪᓗ

ᑕᒪᓐᓇ ᑕᑯᓐᓇᕐᑕᐅᔪᖅ ᓄᓇᓕᖕᒥᐅᑕᓄᑦ ᓈᒻᒪᒃᑲᓗᐊᕐᒪᖔᑦ ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓴᐳᔾᔭᐅᑎᐊᓖᑦ ᐃᓂᖓᓂ ᖃᓄᐃᓕᖓᓂᐊᕐᓂᖏᓐᓂᒃ ᐊᕐᕌᒍᖑᕐᐸ 15-20 ᖑᓕᕈᓂ.

ᐱᓕᕆᐊᒃᓴᐃᑦ ᑲᔪᓯᔪᑦ

- ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔨᖏᑦ ᐱᓕᕆᐊᖃᕐᑐᑦ ᐋᖀᒋᐊᕐᓗᑎᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᓂᒃ ᑐᓴᕐᑕᐅᔪᑦ ᑐᓐᖓᕕᒋᓪᓗᒋᑦ ᑕᐃᑲᓐᓇᓂᒃ ᓇᑦᑎᐊᑦ ᑕᕿᖓᓂ 2024 ᐊᕐᕌᒍᒥ ᓄᓇᓕᖕᓂᒃ ᑐᓴᕐᑎᓪᓗᒋᑦ. ᐱᔭᕇᕐᐸᑦ ᐱᓕᕆᐊᒃᓴᐃᑦ ᐅᕝᕙᐅᒐᑦ:
- ᐊᒥᕋᐃᔭᕐᕕᒃ/ᓯᑯᕝᕕᒃ 2024: ᖃᓇᑕᐅᔭᒃᑯᑦ ᑲᑎᒪᓂᖅᑲᕐᓗᑎᒃ ᓄᓇᓕᖕᒥᐅᓂᒃ ᓱᓕᒑᓗᐊᕐᒪᖔᑕ ᐊᒻᒪᓗ ᐱᔭᕇᕐᓗᒍ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑕᐅᓚᐅᒃᑐᑦ.
- ᑲᑕᒑᓇᕝᕕᒃ/ᐋᒡᔪᓕᕐᕕᐅᑉ ᑕᕿᖏᓐᓂᒃ ᐊᕐᕌᒍ 2024: ᑎᒍᒥᐊᕐᑏᑦ ᐊᒻᒪ ᑭᑐᒥᐊᓂᒃ ᑐᓴᕋᓱᐊᕐᓃᑦ ᐸᕐᓇᒃᑕᐅᔪᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᐅᓚᐅᒃᑐᒥ.
- ᑕᑦᖀᓇᕐᔪᐊᑦ/ᐊᕗᓐᓂᕕᐅᑉ ᑕᕿᐊᓂᒃ ᐊᕐᕌᒍ 2025: ᐱᔭᕇᑕᐅᕙᓂᓪᓕᐊᔪᑦ ᑎᑎᖃᑎᒍ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᒃᑕᐅᓚᐅᒃᑐᒥ ᐊᒻᒪᓗ ᐊᓂᒍᕐᑎᑕᐅᓗᑎᒃ
- ᓇᒻᒪᒋᔭᐅᓂᖏᓐᓂᒃ ᓂᕆᐅᖕᓂᖅ ᑕᕿᖅ ᓇᑦᑎᐊ ᐊᕐᕌᒍ 2025–ᒥ: ᓈᒻᒪᒋᔭᐅᒋᐊᓖᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦ ᑲᑎᒪᔨᖏᓐᓂ ᐊᓂᒍᕐᑎᑕᐅᓗᑎᒃ.

- ᓈᒻᒪᒋᔭᐅᓂᖏᓐᓂᒃ ᓂᕆᐅᖕᓂᖅ ᑕᕿᖅ ᒪᓐᓅᑦ ᐊᕐᕌᒍ 2025-ᒥ: ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᒃᑕᐅᓚᐅᒃᑐᒥ ᓈᒻᒪᒋᔭᐅᒋᐊᓖᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᓂᒍᕐᑎᑕᐅᓗᑎᒃ.

ᐊᐱᖁᑏᑦ ᐅᖃᐅᓯᒃᓴᐃᓪᓘᓐᓃᑦ

ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ ᐅᖃᐅᓯᒃᓴᖃᕈᑎᒃ, ᐃᓱᒫᓘᑎᖃᕈᑎᒃ ᐊᐱᖁᑎᖃᕈᑎᒃᓘᓐᓃᑦ, ᐸᕐᓇᐃᔩᑦ ᑲᑎᒪᔨᖏᑦ ᐊᑐᐃᓐᓇᐅᔪᑦ ᐅᖃᖃᑎᖃᕈᓐᓇᖅᑐᑦ ᖃᖓᑐᐃᓐᓇᖅ.

ᑲᓇᑕᒥ ᒥᕐᖑᐃᕐᓯᕐᕕᓕᕆᔩ: ᓚᕌᓐᑦ ᔫᓇᑦ, laurent.jonart@pc.gc.ca
ᓄᓇᕗᑦ ᒐᕙᒪᑦ: ᐳᕋᑦᓕ ᐱᕆ, BPirie@gov.nu.ca
ᕿᑭᖅᑕᓂ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᑦ: ᔫᕙᓐ ᓯᒥᒃ, jsimic@qia.ca

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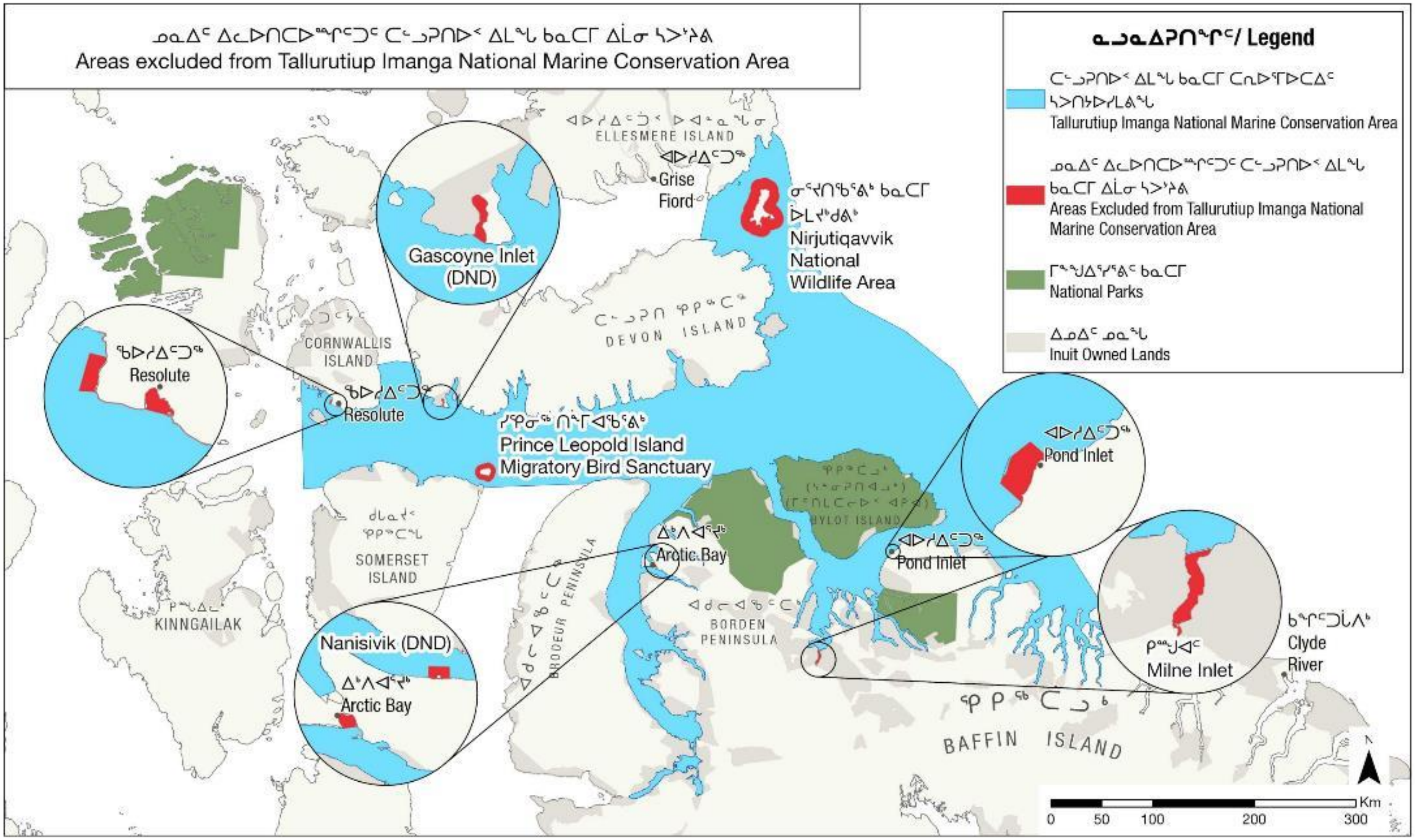
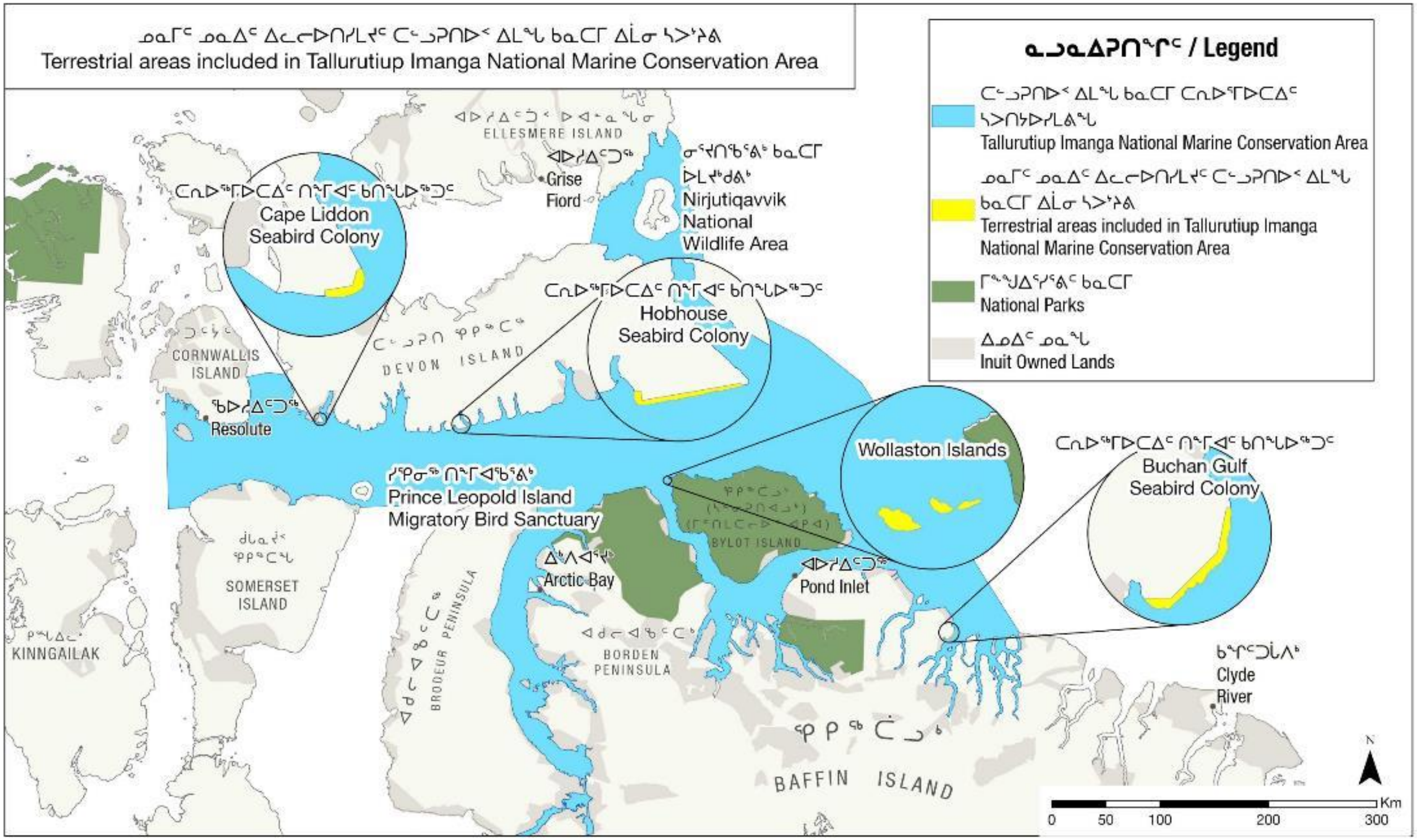
ᓄᓇᐃᑦ ᐃᓚᐅᑎᑕᐅᙱᑦᑐᑦ ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᐃᒫᓂ ᓴᐳᒻᒥᕝᕕ
Areas excluded from Tallurutiup Imanga National Marine Conservation Area





ᓄᓇᒥᑦ ᓄᓇᐃᑦ ᐃᓚᓕᐅᑎᓯᒪᔪᑦ ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᑲᓇᑕᒥ ᐃᒫᓂ ᓴᐳᔾᔨᕕ
Terrestrial areas included in Tallurutiup Imanga National Marine Conservation Area